Influence of Forest Fire on Methyl Mercury in Macroinvertebrate and Fish Communities on the Boreal Plain

by

Erik Webster Allen

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ABSTRACT

Recent studies suggest that logging can increase methyl mercury (MeHg) concentrations in aquatic biota. To determine if forest fire has a similar effect, I studied MeHg in macroinvertebrates and fish from 5 burned and 7 reference lakes on Alberta's Boreal Plain. Two years post-fire, MeHg concentrations in aquatic biota were similar between burned and reference lakes. MeHg concentrations were inversely correlated with lake water pH, total phosphorus concentration, and hardness, reflecting an elevational gradient in lake productivity. A second study was initiated when fire interrupted a logging experiment in a reference watershed. Three months post-fire, MeHg concentrations in biota decreased by 1.3- to 1.5-fold from pre-disturbance concentrations. Lake water concentrations of dissolved organic carbon and inorganic nitrogen increased by 1.2- and 10-fold respectively. Over the short-term, forest fire does not appear to increase MeHg concentrations in aquatic biota on the Boreal Plain.

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Chapter 1. GENERAL INTRODUCTION

1.1 Mercury in aquatic ecosystems

Mercury (Hg) is a neurotoxicant that bioaccumulates in aquatic ecosystems. Fish accumulate Hg as methyl mercury (MeHg), a persistent compound that biomagnifies in lake food webs (e.g., Becker and Bigham 1995). Hg is methylated through microbial and abiotic processes in aquatic and terrestrial environments (Jensen and Jernelov 1969; Nagase 1984). The lipophilic nature of MeHg allows it to readily pass through cell membranes and bioconcentrate in the tissues of aquatic organisms (Mason et al. 1995).

Consumption of fish contaminated with MeHg poses a significant threat to human health. The toxic effects of Hg range from mild to severe neuro-muscular and cognitive dysfunction, and are known collectively as Minamata disease, after a Hg poisoning epidemic in Minamata City, Japan (Harada 1976). The symptoms of Hg poisoning develop when physiological processes such as enzyme function are disrupted by sorption of Hg to sulf-hydryl (SH) groups in proteins (World Health Organization 1991). High Hg concentrations in human populations from northern Canada (Wheatley and Paradis 1995), the Florida Everglades (Fleming et al. 1995), Papua-New Guinea (Abe et al. 1995) and the Brazilian Amazon (Akagi et al. 1995) are linked to fish consumption, and reflect the global impact of this pollutant.

Many remote lakes in Canada's boreal forest contain fish with Hg concentrations that exceed the consumption advisory limit of 0.5 $\mu g \cdot g^{-1}$ (Wren et al. 1991; Bodaly et al. 1993; Garcia and Carignan 2000). The Hg problem in remote freshwater ecosystems is attributed to recent increases in industrial emissions and subsequent atmospheric deposition on lake surfaces and watersheds (Swain et al. 1992; Louchouarn et al. 1993). Hg deposited on boreal catchments may adsorb to terrestrial vegetation or reach soil organic matter via throughfall and litterfall (Hultberg et al. 1994). Hg binds strongly to SH groups in organic matter and accumulates in the humic layer of boreal soils (Johansson and Iverfeldt 1994). Mercury is mobilized from boreal catchments by leaching of dissolved organic carbon (DOC) from humic soil horizons (Lee and Iverfeldt 1991; Meili 1991; Mierle and Ingram 1991). Thus atmospheric deposition and catchment runoff are the primary external sources of Hg to remote aquatic ecosystems.

1.2 Influence of environmental factors on MeHg

The environmental fate of Hg is governed by a complex biogeochemical cycle. Bioaccumulation of Hg in lake food webs is ultimately dependent on external factors that control the flux of Hg and MeHg to lakes, and internal factors (i.e., water chemistry and food web structure) that influence the production, bioavailability, and biomagnification of MeHg.

Watershed characteristics such as drainage ratio (i.e., catchment area:lake area) and peatland cover influence the export of DOC-bound Hg and MeHg to lakes. Drainage ratios have been positively correlated with lake water DOC concentrations (Rasmussen et al. 1989; D'Arcy and Carignan 1997) and lake water MeHg fractions (Lee and Iverfeldt (1991). Peatlands can increase organic carbon and total Hg (THg) in streams (Kolka et al. 1999), DOC and acidity in lakes (Halsey et al. 1997), and act as sources of MeHg to lakes (St. Louis et al. 1994). Where atmospheric deposition and catchment runoff are high, lake inputs of MeHg from these sources can account for the annual bioaccumulation of MeHg by fish (Hultberg et al. 1994). In seepage lakes (i.e., no inflows or outflows, low drainage ratio) and clear water lakes, direct deposition of Hg species to the lake surface and in-lake methylation may be more important than terrestrial sources of Hg and MeHg (Miskimmin et al. 1992; Rudd 1995).

Bioaccumulation of MeHg in lakes is regulated to a large extent by water chemistry. Earlier studies have reported negative correlations between lake water pH and MeHg concentrations in the water column (Watras et al. 1995), zooplankton (Westcott and Kalff 1996; Garcia and Carignan 1999), and fish (Wren et al. 1991; Greenfield et al. 2001). Low pH promotes Hg methylation rates (Xun et al. 1987; Miskimmin et al. 1992) and increases the bioavailability of MeHg in the water column by releasing MeHg bound in sediments or organic particles in the water column (e.g., Miller and Akagi 1979). Watras and Bloom (1992) reported a 3-fold increase in MeHg concentration in phytoplankton following acidification of a lake basin from a pH of 6.1 to 4.7. Other studies, however, report contradictory effects of low pH on Hg partitioning, such as increased sorption of Hg to organic matter (Schindler et al. 1980) and decreased

methylation rates (Ramlal et al. 1985). Site-specific effects and interactions with other lake water constituents likely complicate the relationships between pH and Hg.

The concentration of DOC in the water column also appears to influence MeHg bioaccumulation. In regions where lake water DOC concentrations are low (<10 $mg \cdot L^{-1}$) and of allochthonous origin, MeHg concentrations in fish are positively correlated with DOC (e.g., Wren et al. 1991). Positive correlations between DOC and MeHg likely reflect terrestrial input of DOC-Hg complexes, and associated physico-chemical conditions such as low pH and high colour. High lake water colour appears to reduce the photodegradation of MeHg (Sellers et al. 1996). MeHg concentrations in fish are negatively correlated with DOC where it occurs at high concentrations, such as in seepage lakes (Grieb et al. 1990). This may reflect reduced bioavailability and methylation rates associated with sorption of Hg and MeHg to DOC (Miskimmin et al. 1992; Barkay et al. 1997). Qualitative differences between autochthonous and allochthonous DOC may affect their interactions with MeHg, however the nature of these relationships remains poorly understood.

Bioaccumulation of MeHg and other lipophilic contaminants is reduced in lakes with high productivity (e.g., Taylor et al. 1991; Larsson et al. 1992; Kidd et al. 1999). Several mechanisms have been proposed to explain low MeHg concentrations in productive lakes. High alkalinity, autochthonous DOC, and cation concentrations in productive waters may reduce the uptake of Hg (Rodgers and Beamish 1983) or its toxicity (Pagenkopf 1983). High phytoplankton biomass and growth rates in fish from productive lakes appear to reduce MeHg concentrations through growth biodilution (Larsson et al. 1992; Kidd et al. 1999). Thus MeHg concentrations in aquatic biota are determined by the interactive effects of chemical, ecological, and physical lake and watershed characteristics.

1.3 Biomagnification of MeHg

MeHg concentrations are considerably higher in aquatic biota than in the water column, and increase with trophic position (e.g., Becker and Bigham 1995). MeHg in the water column is bioconcentrated by several orders of magnitude via passive diffusion into phytoplankton (Mason et al. 1995), and biomagnified by 5 to 15-fold at each successive trophic level in the food chain (e.g., Becker and Bigham 1995; Plourde et al. 1997).

Inorganic Hg does not bioaccumulate (Mason et al. 1994), therefore the ratio of MeHg to inorganic Hg also increases with trophic position (e.g., Becker and Bigham 1995). Biomagnification of MeHg in pelagic food chains has been quantified using stable isotope ratios of nitrogen ($\delta^{15}N$) as a time-integrated measure of trophic position (Peterson and Fry 1987; Kidd et al. 1995; Atwell et al. 1998; Bowles et al. 2001). Recent studies indicate that lake-to-lake variation in food chain length can cause significant variation in MeHg concentration among top predators (Cabana et al. 1994).

Uptake and biomagnification of MeHg in the littoral food web is less well understood than for pelagic systems. Few studies have correlated water chemistry with MeHg concentrations in littoral macroinvertebrates, or examined the transfer of MeHg from littoral carbon sources such as sediment, detritus, and macrophytes through primary consumers, detritivores, and fish. Previous studies indicate that MeHg concentrations are higher in predatory macroinvertebrates than in detritivores and herbivores (Tremblay et al. 1996; Hall et al. 1998), however biomagnification of MeHg in the littoral food web has yet to be thoroughly analyzed with stable isotope ratios.

1.4 Influence of watershed disturbance on MeHg

Recent studies from the Boreal Shield suggest that clear-cutting can increase MeHg concentrations in zooplankton and fish (Garcia and Carignan 1999, 2000). The same studies reported no difference between MeHg concentrations in lakes from burned and reference watersheds. Watershed disturbance may affect MeHg concentrations in several ways. First, removal of forest cover can increase erosion, water yield, and export of organic matter (Tiedemann et al. 1978; Bosch and Hewlett 1982; McEachern et al. 2000). These processes may increase the input of DOC-bound Hg to surface waters (e.g., Garcia and Carignan 1999). Second, decreases in lake water pH and light penetration from the input of humic substances may increase net methylation rates (Miskimmin et al. 1992) and reduce photodegradation of MeHg (Sellers et al. 1996), respectively. Post-disturbance increases in lake colour may also reduce biodilution of the MeHg pool by depressing primary productivity (e.g., McEachern et al. 2000). Alternatively, post-fire input of nutrients may increase primary productivity (Carignan et al. 2000) and dilute MeHg in the food web.

The short-term impacts of forest fire and logging on surface water quality in boreal watersheds appear to vary across regions. McEachern et al. (2000) reported higher lake water concentrations of DOC and colour in burned watersheds compared to unburned watersheds in the Boreal Sub-Arctic region. In contrast, Carignan et al. (2000) did not detect an increase in DOC concentration or colour in burned lakes on the Boreal Shield. Prepas et al. (2001a) detected an increase in concentrations of total phosphorus (TP) and algal toxins after logging in productive lakes on Alberta's Boreal Plain, but no increase in DOC concentration or colour, as reported from logged watersheds on the Boreal Shield (Carignan et al. 2000). Regional differences in the impacts of watershed disturbance on surface water chemistry appear to reflect geographic variation in climate, geology, and limnology across the Boreal Forest. Previous studies on the impact of watershed disturbance on MeHg are limited to lakes in one region of the eastern Boreal Shield, and may not extrapolate to other boreal regions, such as the western Boreal Plain.

1.5 Mercury on the Boreal Plain

The Boreal Plain sub-region extends from Manitoba to northern British Colombia and accounts for 25% of Canada's boreal forest (Strong and Leggat 1992). Unlike the Precambrian Shield, the Boreal Plain subregion is characterized by sedimentary geology, deep glacial till, a moderately dry climate, and a varied topography that ranges from low-lying plains to steeply sloped foothills and plateaus (Strong and Leggat 1992). The trophic status of Boreal Plain lakes is highly variable, ranging from meso-oligotrophic to hyper-eutrophic (Mitchell and Prepas 1990). Variability in lake chemistry has been attributed to the influence of landscape features such as wetlands (Halsey et al. 1997), drainage ratio (Prepas et al. 2001b), and groundwater inputs (Shaw et al. 1990). Earlier studies on Boreal Shield lakes have developed models that predict MeHg concentrations based on lake chemistry (Wren et al. 1991) and trophic ecology (Kidd et al. 1995), however, given limnological differences between boreal regions, these relationships may not extrapolate to the Boreal Plain.

Few studies have addressed the MeHg problem in Boreal Plain lakes. Many of Alberta's boreal lakes and rivers have fish consumption advisories due to high concentrations of MeHg, however local aspects of its biogeochemical cycle are poorly

understood (Alberta Environment 2000). Potential sources of Hg in Alberta include naturally occurring Hg in soils (Jonasson and Boyle 1972), reservoir formation (Alberta Environment 1984), hazardous waste incineration (Penner and Associates 1990), and the fossil fuel industry (Donald et al. 1996). In 2000, Alberta's coal-fired power plants released 1 020 kg of Hg to the atmosphere (National Pollutant Release Inventory 2000), approximately 40% of the total Hg released by Canada's coal-fired electricity sector. Recent proposed expansion of thermal generating stations in Alberta has lead to concern over the impacts of increased Hg emissions on aquatic ecosystems.

1.6 Scope of the project

Logging in Alberta's boreal forests has increased three-fold since 1970 (Canadian Council of Forest Ministers 1997), raising concerns over the effects of watershed disturbance on lakes and streams. As a sustainable disturbance, forest fire can provide a natural context for the effects of forest cover removal on aquatic ecosystems (e.g., McEachern et al. 2000). Previously, these effects were thought to be limited to increased water flow and decreased water quality, however recent studies suggest that logging, but not fire, can increase MeHg concentrations in lake food webs (Garcia and Carignan 1999, 2000).

In May 1998, a forest fire burned 150 000 ha of boreal forest near the town of Swan Hills, in northern Alberta. Several upland-dominated, headwater catchments were burned, providing an opportunity to determine the impacts of forest fire on Boreal Plain lakes. A joint study with Environment Canada, Alberta Health and Wellness, and the University of Alberta was initiated to compare the impacts of natural and anthropogenic forest cover removal on MeHg concentrations in aquatic biota. To assess the impact of logging on MeHg, a watershed was selected for prescribed timber harvest (Delorme Lake) with the assistance of industrial partners Millar Western Forest Products Limited, Blue Ridge (1981) Ltd. and Vanderwell Contractors. Half the watershed was logged during winter 2001, following pre-disturbance sample collection in summer 2000, however the experiment was interrupted by a forest fire in spring 2001 that burned the entire watershed. Thus I used two approaches to determine the short-term impacts of forest fire on MeHg and lake chemistry: 1) a pre- vs. post-disturbance comparison in

Delorme Lake, and 2) a comparison between lakes burned in the 1998 Swan Hills fire (2 yr post-fire) and nearby reference lakes.

The Swan Hills has been a region of toxicological concern since 1987, when the Swan Hills Treatment Centre (SHTC) began incineration and storage of hazardous wastes (Stanley and Associates Engineering 1990). Although high concentrations of Hg have been detected in fish from two nearby lakes (Edith and Chrystina), they have not been directly linked to the SHTC. Edith Lake is 10 km west (upwind) from the SHTC, whereas Chrystina Lake is 1.5 km to the north. Neither lake is located in the same drainage basin as the SHTC. As of 1990, deposition of Hg near the facility did not exceed background values (Stanley and Associates Engineering 1990). Edith and Chrystina lakes fit the selection criteria for the study, thus their MeHg concentrations were compared to burned and undisturbed lakes remote from the SHTC.

The study area spanned an elevational gradient and included two distinct ecoregions within the Boreal Plain, the Boreal Foothills and the Boreal Mixedwood (Strong and Leggat 1992). Consequently, my study lakes were characterized by high spatial variability in watershed characteristics. With data from an earlier study on Boreal Plain lakes (Prepas et al. 2001b) I determined the relative influences of watershed characteristics and forest fire on water chemistry (Chapter 2). This analysis provided a context for the interpretation of MeHg concentrations. To determine the impact of forest fire on MeHg (Chapter 3), I focused on the littoral macroinvertebrate and fish communities that dominate shallow Boreal Plain lakes (Beaudoin et al. 2001). As a further objective, I examined the relationships between MeHg concentration, water chemistry, and food web structure. Stable isotopes of nitrogen ($\delta^{15}N$) and carbon ($\delta^{13}C$) were used to delineate trophic position and food source, respectively (Peterson and Fry 1987). This project represents the first comprehensive study of the linkages between MeHg in aquatic biota, watershed disturbance, lake chemistry, and trophic position in Boreal Plain lakes.

This thesis was prepared in paper format. The first paper (Chapter 2) is entitled: "Lake water chemistry of burned and undisturbed watersheds on the Boreal Plain: An ecoregion approach". The second paper (Chapter 3) is entitled: "Influence of forest fire on methyl mercury in macroinvertebrate and fish communities on the Boreal Plain".

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Chapter 2. LAKE WATER CHEMISTRY OF BURNED AND UNDISTURBED WATERSHEDS ON THE BOREAL PLAIN: AN ECOREGION APPROACH[1]

2.1 Introduction

Fire is the dominant natural disturbance in the Boreal Forest (Canadian Council of Forest Ministers 1997). As a sustainable disturbance, forest fire provides an opportunity to determine the impact of large-scale canopy removal on ecosystems and assess the feasibility of applying natural disturbance patterns to forest management (Carignan and Steedman 2000). Recent expansion of the forestry industry in Alberta has led to concern over the effects of watershed disturbance on the surface water quality and biota of lakes in the western Boreal Plain subregion of the Boreal Forest.

Many studies link forest fire with increases in water and nutrient export from streams in forested catchments (Wright 1976; Tiedemann et al. 1978; Bosch and Hewlett 1982; Bayley et al. 1992), however only a small number have considered the impact of fire on the water chemistry of boreal lakes (Wright 1976; Carignan et al. 2000; McEachern et al. 2000). Short-term (i.e., 2 to 3 yr) post-fire increases in total phosphorus (TP) concentrations and primary productivity seem to occur in some lakes (Carignan et al. 2000; McEachern et al. 2000), but not in others (Wright 1976; Bayley et al. 1992). Likewise, post-fire increases in lake water concentrations of dissolved organic carbon (DOC) concentration, colour, and light extinction occur in some regions (McEachern et al. 2000) but not in others (Carignan et al. 2000). The impact of forest fire on surface water chemistry is dependent on regional characteristics such as catchment morphometry, soils, vegetation, climate, and extent of watershed disturbance. Whereas previous studies have focused on nutrient-poor Boreal Shield (Carignan et al. 2000) and peatland-dominated Boreal Sub-Arctic watersheds (McEachern et al. 2000), none have considered the impact of fire on lakes in nutrient-rich, upland-dominated watersheds such as those on the western Boreal Plain.

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The present study focuses on burned watersheds in one of several conifer-dominated upland regions (Boreal Foothills; Strong and Leggat 1992) that punctuate the predominantly low-relief Boreal Mixedwood ecoregion of the Boreal Plain. Due to the eutrophic condition of many Boreal Plain lakes (Mitchell and Prepas 1990) and the high P content of soil porewater (Evans et al. 2000), modest shifts in nutrient dynamics following forest disturbance can cause severe water quality problems (Prepas et al. 2001a). Earlier studies have characterized the limnology and lake-watershed interactions of the Boreal Mixedwood (Prepas et al. 2001a, 2001b), however it is not known whether Boreal Foothills lakes have similar water chemistry and lake-watershed dynamics.

Many of the geographic features that differ between the Foothills and Mixedwood ecoregions, such as elevation, climate, and vegetation, are known to influence surface water chemistry. Foothills lakes are positioned relatively high in the landscape relative to Mixedwood lakes, and may have lower cation concentrations because they are less influenced by the regional groundwater flow (Webster et al. 1996; Kratz et al. 1997). The nutrient content of lake water will also reflect surface and sub-surface runoff, which will have higher DOC and lower TP concentrations under conifer cover in the Foothills (e.g., Cronan and Aiken 1985) than in Mixedwood watersheds (Xiao et al. 1991). Precipitation and runoff are higher in the Foothills relative to lower-elevation Mixedwood regions (Strong and Leggat 1992), thus the potential for post-disturbance soil erosion is greater. These factors may cause Foothills lakes to have a dilute water chemistry and increased susceptibility to the effects of watershed disturbance, relative to lakes in Mixedwood watersheds.

Research on watershed disturbance often relies on a comparative approach where undisturbed lakes are compared to logged or burned lakes. Spatial variability of landscape features and water chemistry can obscure or enhance differences between burned and reference lakes and lead to misinterpretation of data. This problem is magnified in regions such as the Boreal Plain where factors such as drainage ratio and relative importance of internal loading and groundwater to lake water P budgets influence water chemistry (Shaw et al. 1990; Prepas et al. 2001b). Carignan et al. (2000) accounted for variation in catchment morphometry on the Boreal Shield by using impact ratios (burned or logged area:lake volume or lake area) to model nutrient concentrations in 38

lakes. This approach demonstrated that post-disturbance changes in lake water concentrations of TP, DOC, and dominant ions are dependent on the interaction between catchment morphometry and the proportion of the watershed area affected by fire or logging.

As part of the field program of the Forest Riparian and Watershed Disturbance (FORWARD) project (Smith et al. *ms. submitted*), I sampled burned and reference lakes across a range of elevations in Boreal Foothills and Boreal Mixedwood watersheds of northern Alberta. To determine the short-term impact of forest fire I compared the lake water chemistry of burned and unburned watersheds, and tested for correlations between burn impact ratios and water chemistry. To distinguish between fire and ecoregion effects, I used data from previously studied Mixedwood lakes (Prepas et al. 2001b) to compare the water chemistry of lakes from Foothills and Mixedwood ecoregions, and tested for relationships between water chemistry and landscape features.

2.2 Materials and Methods

2.2.1 Study area

The study lakes are located in the Swan Hills, 200 km northwest of Edmonton, Alberta (Fig. 2-1). The region is characterized by steep slopes and plateaus that rise from 600 m to 1 350 m above sea level (Strong and Leggat 1992). The Swan Hills Upland is an erosional remnant comprised of Cretaceous and Tertiary sandstones, shales, clays, and gravels. Repeated glaciations have left deposits of glacial material (ground moraine) that range in thickness from 15 m in upland areas to 30 m along major river valleys, and underlie Orthic Gray Luvisolic soils. Groundwater chemistry in the Swan Hills is dominated by K^+, Na^+, Ca^{2+}, and carbonates (HCO_3^-, CO_3^{2-})(Alberta Research Council 1977). Major drainages in the Swan Hills are the Sakwatamau, Freeman, and Saulteaux rivers, tributaries of the Athabasca River. Boreal and sub-alpine vegetation occur at upper elevations, including lodgepole pine (*Pinus contorta*), white spruce (*Picea glauca*), balsam fir (*Abies balsamea*), and black spruce (*Picea mariana*) (Moss and Pegg 1963). Trembling aspen (*Populus tremuloides*), balsam poplar (*Populus balsamifera*), and white spruce dominate well-drained sites at lower elevations. Peatlands (primarily fens) occur

over ~5% of the landscape and are vegetated by black spruce, tamarack (*Larix laricina*), and sedge species (*Carex spp.*).

The study area straddled two defined ecoregions within the Boreal Plain subregion: the Boreal Foothills (upper elevations) and the Boreal Mixedwood (lower elevations; Fig. 2-1; Strong and Leggat 1992). Boreal Mixedwood and Foothills ecoregions have continental climates, with comparable mean summer temperatures (May to September; 10 and 11.5°C, respectively), but colder winters in the Mixedwood than in the Foothills (December to February; -15 and -10°C, respectively). The Boreal Foothills ecoregion has higher annual precipitation (~610 mm, 1941 to 1970) than the Mixedwood region (~470 mm), and receives a greater proportion of annual precipitation during the winter period of October to April (Strong and Leggat 1992). Local differences in long-term summer rainfall (May to August, 1945 to 1990) in the Swan Hills reflect the range in elevation from the Athabasca River valley at Whitecourt (rainfall 340 mm; elevation 540 m) to the Swan Hills summit at Goose Mountain (458 mm, 1 400 m). The long-term evaporation rate for the region is 525 $mm \cdot yr^{-1}$ (1972 to 1997; Canadian Climate Centre, Data Management Division, Downsview, ON).

In May 1998, a forest fire burned > 150 000 ha of boreal forest in the Swan Hills, including the watersheds of several streams and lakes (Fig. 2-1). Five burned-Foothills (mean disturbance = 62%), four reference-Foothills, and three reference-Mixedwood lakes were selected based on the following criteria: 1) headwater lakes with upland-dominated watersheds (<25% wetlands), 2) 10 to 40 ha in size, and 3) drainage ratio (DR, drainage basin area (DBA)/lake area (LA)) >2. Lake elevations ranged from 610 to 1 115 m. The lakes were shallow, ranging from 1 to 3.9 m in mean depth, with 9±1% wetland area (mean±standard error). Two burned Foothills, three reference Foothills, and two reference Mixedwood lakes were thermally stratified during the sampling period (July to August). Twin Lake (burned) and Goodwin (reference Mixedwood) each had a small (< 6 ha) waterbody that drained into them. Similar to most of the forested portions of the western Boreal Plain, all watersheds had linear disturbances (i.e., cutlines, roads) affecting 1 to 2% of the drainage basin area (DBA), and Mons and Edith lakes each had an oil well located in their watersheds. Edith and Chrystina lakes were located within 10 km of the Swan Hills Treatment Centre.

Ten additional Boreal Mixedwood lakes with upland-dominated watersheds (wetland cover = 16±2%) were selected for analysis from Prepas et al. (2001b). The lakes were located approximately 250 km north (N24 and N29) and 100 to 200 km east (N11, N20, N31, N32, N33, N34, N35 and N37) of the Swan Hills. Data presented for these Boreal Mixedwood lakes were from samples collected in 1996; see Prepas et al. (2001b) for a complete description of the study area.

2.2.2 Lake sampling and analysis

Sampling and analyses followed the methods outlined in Prepas et al. (2001b). The study lakes were sampled once in each of July and August 2000. Profiles of temperature and light penetration (LI-COR 192SA underwater quantum sensor with model LI-1000 datalogger) were measured at one of two sampling sites established near the deepest site in each lake. A Hanna HI9025C pH meter was used to measure pH in the field. Composite euphotic-zone (depth to 1% of surface light) water samples were collected from two sites per lake with weighted Tygon tubing fitted with a one-way valve. Water samples were filtered within 2 h of collection for total dissolved P (TDP), colour, dissolved inorganic nitrogen (DIN; nitrate+nitrite and ammonium) (Millipore, pore size 0.45 µm), and chl *a*, DOC, dominant cations (Na^+, K^+, Mg^{2+}, Ca^{2+}), and anions (Cl^-, SO_4^{-2}) (Gelman G/C filter pore size 1 µm). Chl *a* and DIN samples were frozen immediately on dry ice.

All samples were returned to the Limnology Service Unit at the University of Alberta (Edmonton Alberta) within 72 h of collection. Conductivity was measured on a CDM 83 conductivity meter. Total alkalinity (bicarbonate + carbonate, $mg \cdot L^{-1}$) was determined with a Mettler DL autotitrator (American Public Health Association 1992). Samples for TP and TDP were analyzed with the modified potassium persulfate technique (Prepas and Rigler 1982). Particulate P (PP) was calculated as the difference between TP and TDP. Chlorophyll *a* was extracted with 90% ethanol and analyzed following Bergmann and Peters (1980). Nitrate+nitrite and ammonium samples were analyzed with a Technicon Autoanalyzer (Technicon Autoanalyzer II Methods #158-71 and 154-71). A Shimadzu Total Organic Carbon analyzer (TOC-5000 A) was used to measure DOC concentrations, and colour was measured at 440 nm with a Milton Roy

1001 spectrophotometer. Anions were measured with a Dionex 2000I/SP ion chromatograph fitted with an AS4A-Sc high capacity anion exchange column. Dominant cation concentrations were determined with a Perkin Elmer 3300 atomic absorption spectrophotometer.

Bathymetric maps generated from depth soundings were used to calculate lake volume (LV). Lake area, catchment area (CA, LA + DBA), burned area (BA), and vegetation and wetland cover were determined using 1:20 000 aerial photographs, 1:50 000 topographic maps, and a GIS database (ArcView v. 3.1). Volumetric (VIR, BA/LV) and areal (AIR, BA/LA) impact ratios were calculated after Carignan et al. (2000). Drainage basin slope (%SLOPE) was calculated from the average rise/run of 10 to15 transects (CS1, D'Arcy and Carignan 1997), plotted from the lake edge to the watershed boundary.

Summer rainfall (May to August) was monitored at Swan Dive fire tower (1945-1990, 386 mm; elevation 1 272 m), Whitecourt fire tower (376 mm, 1 200 m), and Whitecourt Airport (340 mm, 750 m). Compared to the long-term average, rainfall at these three stations was on average 25% lower in both summers following the fire (1998 and 1999), and 20% higher in 2000. Runoff was calculated from total annual discharge at a permanently gauged stream in the Swan Hills (Sakwatamau C; see Prepas et al. *ms. submitted*). During the three years following the fire, runoff was 53, 115 and 110 $mm \cdot yr^{-1}$, respectively. In 1996, precipitation in Mixedwood watersheds from Prepas et al. (2001b) was 28% higher than the long-term mean (1970-1994), and runoff was ~170% higher than the 25-yr mean (Prepas et al. 2001a). Water residence time (WRT) for the study lakes was estimated from long-term (1972-1997) mean evaporation rate for at the Sakwatamau C site, and mean runoff for 2000 from the Sakwatamau River, following the method outlined in Prepas et al. (2001a). Annual precipitation in the Foothills lakes was extrapolated from summer rainfall at Swan Dive fire tower and annual precipitation at Sakwatamau C, since fire tower data are only collected during the summer months.

2.2.3 Statistical analysis

Water chemistry in burned and unburned lakes was compared within ecoregions (i.e., burned Foothills ($n = 5$) vs. reference Foothills lakes ($n = 4$)) with the Mann-

Whitney *U*-test. The Mann-Whitney *U*-test was used because sample sizes were too small to reliably determine whether or not the data were normally distributed (Sokal and Rohlf 1995). Similarly, I compared lake chemistry between reference Foothills ($n = 4$) and reference Mixedwood lakes ($n = 3$) with the Mann-Whitney *U*-test. Where there were no significant differences, I grouped burned and reference Foothills lakes ($n = 9$) and compared them to a pooled group consisting of reference Mixedwood lakes from the Swan Hills ($n = 3$) and Mixedwood lakes selected from Prepas et al. (2001b) ($n = 10$). Differences in watershed features and water chemistry between pooled Foothills and Mixedwood lakes were determined with the Student's *t*-test. All water chemistry values represent July to August means, the sampling overlap period between the present study and Prepas et al. (2001b). July data was unavailable for two lakes in the Swan Hills (Edith and Delorme) and three lakes (N20, N24, N29) from Prepas et al. (2001b). Only TP, TDP, and chl *a* concentrations showed seasonal fluctuations, thus July values for these parameters were extrapolated based on % change from July to August in lakes from corresponding regions.

Pearson correlation coefficients were calculated to determine covariation among watershed characteristics and water chemistry variables. Linear regression was used to examine relationships between colour and DOC concentrations in Foothills and Mixedwood lakes, and slopes were compared between ecoregions with ANCOVA. All water chemistry and catchment variables were tested for normality with the Shapiro-Wilkes test. Where necessary, non-normal variables were $\log_{10}$ transformed prior to analysis of variance and proportions were arcsine transformed (i.e., %CON, %DEC, %WET, %SLOPE and %burn disturbance (%BURN)). Transformed variables with non-normal distributions were analyzed with non-parametric analyses. All statistical analyses were conducted with SPSS (v. 8.0) and results were considered significant at $P \leq 0.05$.

2.3 Results

Physical characteristics of the Swan Hills watersheds are presented in Table 2-1. Burned and reference Foothills watersheds were similar in all categories including conifer cover (mean±standard error = 77±6 and 88±4%, respectively, Mann-Whitney *U*-test, $P = 0.6$), drainage ratio (CA/LV, 5.2±1.1 and 4.8±2.7 m^{-1}, $P = 0.3$), and elevation

(1042±24 and 1055±29 m, $P = 0.5$). In contrast, reference watersheds in the Mixedwood ecoregion were dominated by deciduous cover (69±13%) and were 30% lower in elevation (748±73 m, $P = 0.03$) than reference Foothills catchments. Lake water residence times were longer in Mixedwood lakes (11.3±1 yr) than in Foothills lakes (5.9±1.4 yr, $P = 0.04$), reflecting relatively high precipitation at upper elevations. Lake trophic status ranged from oligo-mesotrophic to mesotrophic based on euphotic zone concentrations of chl *a* (Mitchell and Prepas 1990). Concentrations of chl *a* in lake water ranged from 2.0 to 19.1 $\mu g \cdot L^{-1}$ (Table 2-3). TP concentrations (range = 16 to 38 $\mu g \cdot g^{-1}$) were strongly correlated to chl *a* concentrations ($r = 0.91$, $P << 0.01$) and pH ($r = 0.78$, $P = 0.003$). Lake water concentrations of DOC (6 to 32 $mg \cdot L^{-1}$) were not correlated with colour (12 to 139 $mg \cdot Pt^{-1}$, $P = 0.9$), suggesting in-lake production of organic carbon. In general, lake productivity, alkalinity, and water hardness appeared to be lower than previously reported from the Boreal Plain (Prepas et al. 2001a), but higher than reported from Boreal Shield lakes (Carignan et al. 2000).

2.3.1 Lake water chemistry of burned watersheds

Burned Foothills lakes had 1.4-fold higher concentrations of DOC, 1.3-fold higher TP, and 1.5-fold higher chl *a* than reference-Foothills lakes (Fig. 2-2), however only DOC concentrations were significantly different (Mann-Whitney *U*-test, $P = 0.05$, 0.1, and 0.09 respectively). Mean pH in reference Foothills lakes (7.0) was similar to burned Foothills lakes (7.3, $P = 0.6$), as were conductivity ($P = 0.5$), alkalinity ($P = 0.3$), and DIN concentrations ($P = 0.3$; Table 2-2). Trends in lake water DOC, TP, and chl *a* concentrations were consistent with post-fire inputs of nutrients reported from earlier studies (e.g., McEachern et al. 2000).

Volumetric and areal impact ratios (VIR, AIR) were correlated with several water chemistry parameters in burned lakes (Table 2-3). Colour in the euphotic zone of burned lakes increased with the area of watershed burned divided by lake area (AIR, $r = 0.90$, $P = 0.03$) and lake volume (VIR, $r = 0.98$, $P = 0.01$). Correlations of colour with DBA/LA and DBA/LV (i.e., removing the disturbance component of the AIR and VIR parameters) were non-detectable ($P = 0.3$ and 0.2, respectively). Further, colour in burned lakes was strongly correlated with DOC concentration (log colour vs. log DOC, $r = 0.93$, $P = 0.02$),

although DOC was not significantly correlated to impact ratios. Light extinction coefficients (m^{-1}, base e) also increased with AIR ($r = 0.94$, $P = 0.02$) and VIR ($r = 0.90$, $P = 0.04$) but were not correlated with drainage ratios (DBA/LA, DBA/LV, $P = 0.2$ and 0.1, respectively). There were also inverse correlations between impact ratios and lake water pH (Fig. 2-3), alkalinity, and conductivity (Table 2-3). Correlations of pH, alkalinity, and conductivity with drainage ratios were non-significant ($P = 0.2$ and 0.5, respectively). These data suggest a combined effect of catchment morphometry and percent disturbance on the export of humic substances from burned catchments in Boreal Foothills lakes.

2.3.2 Ecoregion comparison

Boreal Foothills lakes in the Swan Hills appeared to have a dilute water chemistry relative to Boreal Mixedwood lakes. Reference Mixedwood lakes had 2.1-fold higher concentrations of DOC (Mann Whitney *U*-test, $P = 0.03$), 1.8-fold higher TP ($P = 0.03$) and 4.4-fold higher chl *a* ($P = 0.03$) than reference Foothills lakes (Fig. 2-2). Mean pH was 8.5 in Mixedwood lakes compared to 7.0 in reference Foothills lakes. Reference Mixedwood lakes also had 3.3-fold higher conductivity and 4.2-fold higher alkalinity than reference Foothills lakes ($P = 0.03$ and 0.03, respectively). Lower productivity and water hardness indices in the conifer-dominated Foothills compared to the deciduous-dominated Mixedwood region were reflected in watershed characteristics (Table 2-4). Among all 12 watersheds, conifer cover was negatively correlated with lake water concentrations of TP ($r = 0.63$, $P = 0.03$), alkalinity ($r = -0.7$, $P = 0.01$), chl *a* ($\log_{10}$chl *a*, $r = -0.72$, $P = 0.008$) and K^+ ($\log_{10}K^+$, $r = -0.76$, $P = 0.004$). Lake water DOC concentration was negatively correlated with elevation ($\log_{10}$DOC, $r_s = -0.64$, $P = 0.03$). Water chemistry parameters were not correlated with drainage ratio (CA/LV), mean depth, or wetland cover, therefore the difference in lake chemistry between Mixedwood and Foothills lakes may be attributable to ecoregional effects.

To further examine differences between ecoregions, data from the Swan Hills and previously studied Mixedwood watersheds (Prepas et al. 2001b) were pooled and compared according to ecoregion. Pooled Foothills watersheds ($n = 9$) were 1.6-fold higher in elevation (Mann-Whitney *U* test, $P << 0.01$) and had 9-fold higher conifer

cover than pooled Mixedwood catchments ($n = 13$, $P << 0.01$, Table 2-5). Mixedwood lakes had 2.5-fold larger lake area ($P = 0.02$) and 2.7-fold larger catchments ($P << 0.01$) than Foothills lakes. Important predictive characteristics of water chemistry such as wetland cover and drainage ratio (Prepas et al. 2001b) did not differ between the two ecoregions.

Many of the significant differences in water chemistry between Foothills and Mixedwood lakes in the Swan Hills remained when the lakes were pooled (Fig. 2-4). Mixedwood lakes ($n = 13$) had higher concentrations of TP (2.6-fold, $P << 0.01$), chl *a* (3-fold, $P < 0.004$), conductivity (3.6-fold, $P << 0.001$) and alkalinity (3-fold, $P < 0.01$) than Foothills lakes. High conductivity in the pooled Mixedwood lakes reflected 3.3-fold higher concentrations of $Ca^{2+} + Mg^{2+}$ ($P << 0.01$), 5.3-fold higher K^+ ($P << 0.01$), and 1.9-fold higher Na^+ relative to Foothills lakes ($P = 0.03$, Fig. 2-4). Numerous water chemistry parameters were negatively correlated with elevation including lake water conductivity and TP (Fig. 2-4), chl *a*, alkalinity, $Ca^{2+} + Mg^{2+}$, Na^+, and K^+ concentrations (Table 2-7). High variation in water chemistry at the low elevations suggested a local source of variability in Boreal Mixedwood watersheds (Fig. 2-5).

Whereas mean DOC concentration in pooled Mixedwood lakes ($n = 13$) was 1.8-fold higher than in reference-Foothills lakes ($n = 4$, $P = 0.03$, Fig. 2-4), there was no detectable difference in lake water colour between ecoregions ($P = 0.8$). In the combined 22-lake set, a weak correlation was observed between log CA/LV and log colour ($r = 0.47$, $P = 0.04$), as has been reported previously in Boreal Plain lakes (Prepas et al. 2001b). Separate linear regressions of log colour vs. log DOC indicated a strong relationship in Foothills lakes (log colour = -0.55 + 2.1*log DOC, $r^2 = 0.92$, $P << 0.01$) and a poor relationship in Mixedwood lakes (log colour = 0.53 + 0.86*log DOC, $r^2 = 0.40$, $p = 0.02$, Fig. 2-6). The slope of the Foothills regression line was steeper than that of the Mixedwood ecoregion (ANCOVA, $P = 0.02$). The strong relationship between lake water DOC concentration and colour in the Foothills was likely influenced by increased post-fire export of humic substances in the burned lakes. The poor relationship among Mixedwood lakes may reflect the combined effects of low wetland cover (i.e., < 25%), low conifer cover, and high in-lake DOC production.

2.4 Discussion

2.4.1 Influence of forest fire on lake water chemistry

High concentrations of lake water DOC, together with correlations between colour and impact ratios, suggest a short-term increase in allochthonous carbon inputs to burned Foothills lakes. Previously, post-fire increases in DOC concentration had only been reported for peatland catchments in the Boreal Sub-Arctic (McEachern et al. 2000). On the Boreal Shield, clear-cutting caused increased concentrations of DOC and colour in lakes, whereas forest fire did not (Carignan et al. 2000). Vegetation and organic soil layers in burned Boreal Shield catchments were incinerated by fire in the burned catchments, precluding export of organic carbon (Carignan et al. 2000). Compared with upland watersheds on the Boreal Shield, where a thin layer of soil covers bedrock, the Swan Hills catchments have well-developed soil profiles (Strong and Leggat 1992) and should be more resilient to disturbance from fire. Because the Swan Hills fire occurred in the spring while the ground was frozen, damage to the litter and organic soil layers appears to have been reduced, allowing for increased export of humic substances to burned lakes, relative to unburned catchments. Prepas et al. (*ms. submitted*) observed an increase in export of water from burned relative to reference streams in the Swan Hills. The results of Prepas et al. (*ms. submitted*) indicate that forest fire altered the hydrology of Swan Hills watersheds over the short-term, and provide a mechanism for the differences in water chemistry I observed between burned and reference Foothills lakes.

Assessing the impact of fire on DOC concentrations in Boreal Plain lake water was complicated by high concentrations of autochthonous DOC. Given high TP and chl *a* concentrations in the burned lakes relative to unburned lakes, the difference in DOC concentration may also reflect increased autochthonous production of carbon. DOC concentrations in the study lakes were high (~15 $mg \cdot L^{-1}$) relative to Boreal Shield lakes in upland watersheds (~5 $mg \cdot L^{-1}$; Carignan et al. 2000). Positive correlations with lake water concentrations of chl *a* and TP suggest that a high proportion of DOC in Boreal Plain lake water is internally-produced. Despite the influence of autochthonous carbon (typically less coloured than allochthonous carbon) on DOC concentrations, lake water in both the Mixedwood and Foothills sites produced a linear relationship between colour

and DOC, as reported elsewhere (Rasmussen et al. 1989). The high slope of the DOC vs. colour relationship in Foothills lakes indicates greater influence of allochthonous DOC in coniferous watersheds than in those of the deciduous-dominated Mixedwood ecoregion. Consequently, watershed disturbance in the Boreal Foothills could be expected to have a greater effect on lake water colour and humic content than in the Boreal Mixedwood.

The inverse correlation between relative burned area (impact ratios) and lake water pH may reflect increased export of humic acids from burned, coniferous watersheds in the Boreal Foothills. Post-fire increases in the export of acid anions (e.g., humic acids and SO_4^{-2}) have previously been implicated to decrease lake water pH in boreal watersheds (Brosset 1979; Oliver et al. 1983; Bayley et al. 1992; McEachern et al. 2000). McEachern et al. (2000) saw lower alkalinity and 5-fold higher H^+ concentrations in burned than reference lakes, though peatland systems dominated the catchments and acted as natural sources of protons. I did not detect increases in SO_4^{2-} concentrations in burned lakes, contrary to Carignan et al. (2000) who reported 6-fold higher SO_4^{2-} concentrations in burned relative to reference lakes, but no difference in pH. Carignan et al. (2000) reported a rapid decrease (50% lower by the second year) in concentrations of ions that were affected by fire, thus sampling in the third summer following the Swan Hills fire may have reduced the potential to detect differences in ion chemistry.

Differences in water chemistry observed between burned and reference Foothills lakes in the present study were modest compared with previous studies from burned watersheds in the Boreal Forest region. Mean TP and chl *a* concentrations in burned Foothills lakes were 1.3 and 1.5-fold higher than unburned lakes, however the differences were not significant. Prepas et al. (*ms. submitted*) reported higher PP export in a Swan Hills stream, two years post-fire, compared to a pre-fire year (Munn and Prepas 1986). Three years post-fire, Carignan et al. (2000) saw 2 to 3-fold differences in TP, chl *a* and DIN concentrations in Boreal Shield lake water. Two years post-fire, McEachern et al. (2000) reported 2-fold increases in lake water TP concentration in Boreal Sub-Arctic catchments. Catchments in the Swan Hills were impacted less (mean disturbance = 62%) than previous studies (91 and 83%, respectively, for Carignan et al. 2000, McEachern et al. 2000). The impact ratio models presented by Carignan et al. (2000) indicated that short-term response to fire occurs most readily in shallow lakes with large DR (>4) and

% disturbance (>30% of the DBA). Based on models derived from the Boreal Shield, burned watersheds in the Swan Hills had insufficient disturbance and DRs to affect an increase in lake water TP concentrations that was detectable with my sample size. Factors such as the release of P from lake sediments and groundwater inputs may have further obscured the effect of the Swan Hills fire on P budgets in burned lakes (Riley and Prepas 1984; Shaw et al. 1990).

Previous studies from burned upland stream and lake catchments indicate increased cation concentrations in surface water following disturbance (e.g., Tiedemann et al. 1978; Carignan et al. 2000). There was no evidence of increased export of base cations in the water chemistry of burned Swan Hill lakes. Conversely, lake water conductivity decreased with increasing disturbance ratios. A possible explanation for post-fire dilution of cations is the increased export of water in burned watersheds, as was shown to occur in stream watersheds in the Swan Hills (Prepas et al. *ms. submitted*). Reduced infiltration following forest fire could affect lake hydrology by increasing the influence of surface inputs relative to groundwater, an important source of Ca^{2+} and Mg^{2+} to lakes in sedimentary basins (Webster et al. 1996).

Although Prepas et al. *(ms. submitted)* showed higher than expected water export from a burned watershed 2 years post-fire, the potential for increased nutrient inputs to the burned Foothills lakes was reduced by lower than average precipitation and runoff after the fire. In the first two summers following the fire (1998, 1999), rainfall was 25% below the long-term mean and there were few major storm events to generate runoff. Cooke and Prepas (1998) show that storm events, rather than spring runoff, are responsible for most of the P export from forested watersheds on the Boreal Plain. Mean runoff over the 2 years after the fire was 33% lower than the long-term average (1972 to 1997), and was much lower than runoff in similar sized Boreal Shield watersheds (Carignan et al. 2000). In addition, lower than average precipitation slows flushing rates for lakes and can delay the impacts of surface and subsurface nutrient inputs to lake water. Water residence times for burned lakes in the Swan Hills were >3 yr, whereas those studied by Carignan et al. (2000) and McEachern et al. (2000) had comparatively rapid flushing rates (mean = 1.6, 0.18 yr respectively) and were more likely to reflect watershed disturbance in the short-term. Results from a riparian buffer width study on

the Boreal Plain (Prepas et al. 2001a), indicated that increases in lake water TP concentrations following timber harvesting were most prominent in lakes with short water residence times (i.e., ~1 yr). Longer time-scales may be required to accurately assess the influence of water residence time and climatic variability on post-disturbance fluctuations of TP concentrations in Boreal Plain lakes.

2.4.2 Influence of ecoregional factors on lake water chemistry

Differences in water chemistry between Mixedwood and Foothills lakes suggest ecoregion-specific controls on water chemistry. High conductivity and dominant cation concentrations in the Mixedwood lakes, and negative relationships of these parameters with elevation may indicate higher groundwater inputs relative to the Foothills lakes (Eilers et al. 1983; Webster et al. 1996). In sedimentary basins, lakes located at low elevations, such as the Mixedwood lakes, appear to receive greater inputs of groundwater compared to lakes at relatively high elevations (Kratz et al. 1997). In addition to dominant cations, groundwater may represent an important source of P to Mixedwood lakes (Shaw et al. 1990). A confounding factor in the present study was the difference in vegetation between watersheds at high and low elevations. Soil concentrations of P, Ca^{2+}, and Mg^{2+} in Orthic Gray Luvisols are higher under aspen stands relative to conifer stands on the Boreal Plain (Xiao et al. 1991), and P export rates appear higher for deciduous catchments in the Boreal Mixedwood than for coniferous catchments in the Boreal Foothills (Munn and Prepas 1986; Cooke and Prepas 1998). Higher than average runoff (170% above average) in the Mixedwood catchments in 1996 did not appear to drastically affect TP concentrations in lake water as values from 1995, a low runoff year, did not vary considerably from 1996 values for a set of 12 lakes on the Boreal Plain (SE/mean TP for 1995 and 1996 = ~0.1; Prepas et al. 2001a).

Climate may also contribute to differences in water chemistry between ecoregions. Long-term hydrological data show that lakes in the Boreal Foothills receive more precipitation and runoff than Boreal Mixedwood lakes, where annual lake evaporation exceeds precipitation (Alberta Environment, pers. comm.; Prepas et al. 2001a). Higher potential lake evaporation at lower elevations can concentrate solutes in lakes and streams, as was suggested by a study on the Boreal Shield in which lake solute

concentrations varied over an elevational gradient (D'Arcy and Carignan 1997). Based on estimates from long-term precipitation, Foothills lakes appear to receive ~2-fold more water (per area) through direct precipitation and runoff than lakes in the Mixedwood ecoregion. Thus the combined effects of climate, groundwater inputs, and the influence of vegetation on soil nutrients contributed to water chemistry patterns in the Boreal Foothills and Mixedwood ecoregions of the Boreal Plain.

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Table 2-1. Watershed and lake characteristics.

Lake	Ecoregion	Latitude	Longitude	LA (ha)	CA (ha)	LV (10^4 m^3)	CA/LV (m^{-1})	Z_m (m)	WRT (yr)	ELEV (m)	CON (%)	DEC (%)	WET (%)	SLOPE (%)	BURN (%)
Burned															
Mons	Foothills	54°35.4'	115°46.8'	19	55	21	2.6	1.1	3.7	990	90	0	7	4.0	90
Twin	Foothills	54°39.0'	115°51.6'	13	143	18	7.8	1.4	4.7	1040	88	0	10	2.5	70
Shelly	Foothills	54°40.2'	115°45.6'	23	128	45	2.8	2.0	6.6	1100	57	32	11	5.9	80
Michel	Foothills	54°37.2'	115°38.4'	13	133	18	7.2	1.4	4.7	990	63	23	10	6.6	50
Swartz	Foothills	54°40.8'	115°39.0'	28	328	61	5.4	2.2	7.2	1090	86	4	7	5.2	20
Mean				19	157	33	5.2	1.6	5.4	1042	77	**12**	9	4.8	62
Reference															
Rat	Foothills	54°39.6'	115°34.8'	10	128	10	13.0	1.0	3.3	1095	80	10	9	4.4	
Marigold	Foothills	54°40.2'	115°37.8'	34	96	52	1.8	1.5	5.1	1115	87	0	13	4.4	
Christina	Foothills	54°47.4'	115°11.4'	39	261	153	1.7	3.9	13.1	1015	95	0	4	7.7	
Edith	Foothills	54°48.0'	115°22.8'	22	77	28	2.8	1.3	4.2	995	80	0	17	7.9	
Mean				26	140	61	4.8	1.9	6.4	1055	88	2.5	12	6.1	
Pleasant	Mixwood	54°31.8'	115°27.6'	18	254	37	6.9	2.0	12.4	775	4	94	2	2.0	
Goodwin	Mixwood	54°05.4'	115°54.6'	18	253	39	6.5	2.2	13.1	860	32	64	3	5.2	
Delorme	Mixwood	55°05.4'	114°20.4'	34	285	60	4.8	2.1	8.5	610	40	50	10	1.0	
Mean				23	264	45	6.1	2.1	11.3	**748**	**25**	**69**	5	2.7	
Overall mean				22	178	45	5.2	1.8	7.2	973	67	23	9	4.7	-

Note: LA, lake area; CA, catchment area; LV, lake volume; Z_m, mean depth; WRT, water residence time; ELEV, elevation; CON, % conifer in drainage basin; DEC, deciduous cover; WET, wetland cover; BURN, percent burned; SLOPE, catchment slope. Means in bold are different relative to reference-Foothills means (Mann-Whitney, $P \leq 0.05$).

Table 2-2. July to August means for water quality.

Lake	TP ($\mu g \cdot L^{-1}$)	Chl *a* ($\mu g \cdot L^{-1}$)	DOC ($mg \cdot L^{-1}$)	Colour ($mg\ Pt \cdot L^{-1}$)	DIN ($\mu g \cdot L^{-1}$)	pH	COND ($\mu S \cdot cm^{-1}$)	ALK ($mg \cdot L^{-1}$)	$Ca^{2+}+Mg^{2+}$ ($mg \cdot L^{-1}$)	K^{+} ($mg \cdot L^{-1}$)	Na^{+} ($mg \cdot L^{-1}$)	SO_4^{2-} ($mg \cdot L^{-1}$)	Cl^{-} ($mg \cdot L^{-1}$)
Burned-Foothills													
Mons	28	4.4	12	36	5.5	8.7	82	47	10	0.27	7.1	2.6	0.18
Twin	23	5.6	17	139	6.1	6.8	28	12	6	0.45	1.1	1.4	0.08
Shelly	24	4.8	9	29	9.8	7.5	88	33	11	0.49	1.9	0.3	0.06
Michel	19	4.4	16	71	6.7	7.4	61	33	12	0.75	2.2	0.4	0.07
Swartz	31	7.3	12	40	12.5	8.0	94	69	25	0.83	3.6	4.6	0.11
Mean	25	5.3	**13**	63	8.1	7.3	71	39	13	0.56	3.2	1.9	0.10
Reference-Foothills													
Rat	24	3.7	11	34	8.2	8.1	84	47	16	0.21	2.5	0.8	0.06
Marigold	16	2.0	6	12	18.6	7.7	57	32	11	0.43	1.2	0.7	0.07
Christina	20	5.2	8	17	6.7	7.1	32	17	8	0.82	1.3	0.8	0.09
Edith	16	3.7	11	46	9.7	6.6	64	8.7	2	0.86	1.8	0.3	1.50
Mean	19	3.6	9	27	10.8	7.0	59	26	9	0.58	1.7	0.6	0.43
Reference-Mixedwood													
Pleasant	34	13.8	14	16	8.1	9.4	138	87	26	3.60	2.8	0.4	0.27
Goodwin	32	14.1	12	41	10.1	8.1	202	118	41	1.22	4.4	4.2	0.13
Delorme	38	19.1	32	72	9.5	8.6	246	122	39	2.26	7.3	2.0	0.50
Mean	**35**	**15.7**	**19**	43	9.2	**8.5**	**195**	**109**	**35**	**2.36**	**4.8**	2.2	0.30
Overall mean	25	7.3	13.2	46	9.3	7.8	98	52	19	1.02	3.1	1.5	0.26

Note: TP, total phosphorus; chl *a*, chlorophyll *a*; DOC, dissolved organic carbon; DIN, dissolved inorganic nitrogen, COND, conductivity; ALK, alkalinity. Bold means are different from reference-Foothills means; Mann-Whitney *U*-test, $P \leq 0.05$.

Table 2-3. Pearson correlation coefficients for areal and volumetric impact ratios.

	AIR	VIR
Colour (mg $Pt \cdot L^{-1}$)	0.90*	0.98*
Light extinction coefficient (m^{-1} base e)	0.94*	0.90*
COND ($\mu S \cdot cm^{-1}$)	-0.92*	-0.98*
pH	-0.94*	ns
ALK ($mg \cdot L^{-1}$)	ns	-0.89*

Note: AIR, areal impact ratio (burned area/lake area); VIR, volumetric impact ratio (burned area/lake volume); TP, total phosphorus; COND, conductivity; ALK, alkalinity (*P <0.05; ns, not significant).

Table 2-4. Pearson correlation coefficients for selected watershed characteristics and lake water chemistry variables (n = 12).

	ELEV (m)	CA (ha)	CA/LV (m^{-1})	Z_m (m)	arcsine %CON	%WET
log chl *a* ($\mu g \cdot L^{-1}$)	-0.63*	0.71*	ns	ns	-0.72*	ns
TP ($\mu g \cdot L^{-1}$)	-0.59*	0.68*	ns	ns	-0.63	ns
log COND ($\mu S \cdot cm^{-1}$)	ns	ns	ns	ns	-0.73*	ns
ALK ($mg \cdot L^{-1}$)	ns	0.64*	ns	ns	-0.70*	ns
log DOC ($mg \cdot L^{-1}$)	-0.64*	ns	ns	ns	ns	ns
$Ca^{2+} + Mg^{2+}$ ($mg \cdot L^{-1}$)	ns	0.67*	ns	ns	ns	ns
log K^+ ($mg \cdot L^{-1}$)	-0.69*	0.67*	ns	ns	-0.76**	ns
pH	ns	ns	ns	ns	-0.62*	-0.58*

Note: ELEV, elevation; CA, catchment area; CA/LV, CA/lake volume; Z_m, mean lake depth; %DEC, % decidous cover in the drainage basin; %CON, conifer cover; %WET, wetland cover; Chl *a*, chlorophyll *a*; TP, total phosphorus; COND, conductivity; ALK, alkalinity; DOC, dissolved organic carbon; *$P < 0.05$; **$P < 0.001$; ns, not significant; Spearman's rank correlation coefficient calculated for ELEV.

Table 2-5. Watershed characteristics of Boreal Foothills and Boreal Mixedwood lakes.

Characteristic	Boreal Foothills ecoregion (n = 9)				Boreal Mixedwood ecoregion (n = 13)				
	Mean	*SE*	Min.	Max.	Mean	*SE*	Min.	Max.	*P*
LA (ha)	22	3.3	10	34	56	13	14	170	0.02
CA (ha)	150	29	55	328	401	57	120	727	<0.001
DBA/LA (DR)	6.7	1.4	1.8	12	8.5	1.2	1.5	15	0.32
LV (10^4 m^3)	45	15	10	153	130	30	30	340	0.01
CA/LV (m^{-1})	5	1.2	1.8	13	8.0	1.2	1.7	18	0.96
Z_{max} (m)	4.4	1	1.7	10	6.5	1.1	1.8	11	0.45
Z_m (m)	1.8	0.3	1.0	3.9	2.6	0.2	0.9	4.8	0.14
WRT (yr)	5.8	1	3.3	13	8.3	1.1	3.8	17	0.2
ELEV (m)	1048	17	990	1115	658	22	572	860	<0.001
%CON	82	4	57	90	9.1	2.3	1.2	32	<0.001
%DEC	8	4	0	32	73	4.5	33	85	<0.001
%WET	11	1	7	17	14	2.2	1	25	0.34
%SLOPE	4.4	0.5	2.5	7.9	4.5	0.7	1.0	8	0.99

Note: DBA, drainage basin area; LA, lake area; CA, catchment area (CA = DBA+LA); DR, drainage ratio; Z_{max}, maximum depth; Z_m, mean depth; WRT, water residence time; ELEV, elevation; %CON, conifer cover; %DEC, deciduous cover; %WET, wetland cover, %SLOPE, drainage basin slope. *P* value calculated from Student's *t*-test.

Table 2-6. July to August means for water quality of Boreal Foothills and Boreal Mixedwood lakes.

	Boreal foothills lakes (n = 9)				Boreal mixedwood lakes (n = 13)				
Characteristic	Mean	*SE*	Min.	Max.	Mean	*SE*	Min.	Max.	*P*
TP ($\mu g \cdot L^{-1}$)	19	1.9	16	31	50	5.6	29	98	<0.001
TDP ($\mu g \cdot L^{-1}$)	6.6	0.8	5	9	16	2.3	6.4	29	0.002
Chl *a* ($\mu g \cdot L^{-1}$)	3.6	0.6	2	7.3	16	2.4	2.9	28	<0.001
DOC ($\mu g \cdot L^{-1}$)	8.9	1.2	5.8	17	16	1.8	11	31	0.03
Colour (mg $Pt \cdot L^{-1}$)	27	8	12	139	38	5.4	16	71	0.8
pH	7.0	-	6.8	8.7	7.8	-	7.2	9.4	0.09
COND ($\mu S \cdot cm^{-1}$)	66	8	28	94	235	24	112	411	<0.001
ALK ($mg \cdot L^{-1}$)	33	6.4	12	69	97	8.7	47	141	<0.001
$Ca^{2+} + Mg^{2+}$ ($mg \cdot L^{-1}$)	11	3.3	10	29	36	3.7	15	66	<0.001
K^+ ($mg \cdot L^{-1}$)	0.6	0.1	0.2	0.8	3.2	0.4	1.2	5.9	<0.001
Na^+ ($mg \cdot L^{-1}$)	2.5	0.6	1.1	7.1	4.7	0.6	1.0	7.5	0.03
Cl^- ($mg \cdot L^{-1}$)	0.25	0.16	0.06	0.18	0.86	0.32	0.15	1.94	0.26
SO_4^{2-} ($mg \cdot L^{-1}$)	1.3	0.5	0.3	4.6	8.6	5.8	0.1	74	0.31

Note: TP, total phosphorus; TDP, total dissolved phosphorus; Chl *a*, chlorophyll *a*; DOC, dissolved organic carbon; COND, conductivity; ALK, alkalinity (*P* values calculated from Student's *t*-test).

Table 2-7. Pearson correlation coefficients for selected watershed characteristics and water chemistry variables from Boreal Foothills and Mixedwood lakes (n = 22).

	ELEV (m)	log CA (ha)	log CA/LV	log Z_m (m)	arcsine %CON	arcsine %WET
log chl *a* ($\mu g \cdot L^{-1}$)	-0.59*	ns	ns	ns	-0.50*	ns
TP ($\mu g \cdot L^{-1}$)	-0.75**	ns	ns	ns	-0.59*	ns
COND ($\mu S \cdot cm^{-1}$)	-0.81**	0.50*	ns	ns	-0.77**	ns
ALK ($mg \cdot L^{-1}$)	-0.72**	0.47*	ns	0.57*	-0.66*	ns
log colour ($mg\ Pt \cdot L^{-1}$)	ns	ns	0.47*	ns	ns	ns
$Ca^{2+} + Mg^{2+}$ ($mg \cdot L^{-1}$)	-0.66**	ns	ns	ns	-0.58**	ns
Na^{+} ($mg \cdot L^{-1}$)	-0.56*	ns	ns	ns	ns	ns
K^{+} ($mg \cdot L^{-1}$)	-0.80**	ns	ns	ns	-0.80**	ns

Note: ELEV, elevation; CA, catchment area; CA/LV, CA/lake volume; Z_m, mean lake depth; %DEC, % decidous cover in the drainage basin; %CON, conifer cover; %WET, wetland cover; Chl *a*, chlorophyll *a*; TP, total phosphorus; COND, conductivity; ALK, alkalinity; *$P < 0.05$; **$P < 0.001$; ns; not significant; Spearman's rank correlation coefficient calculated for ELEV.

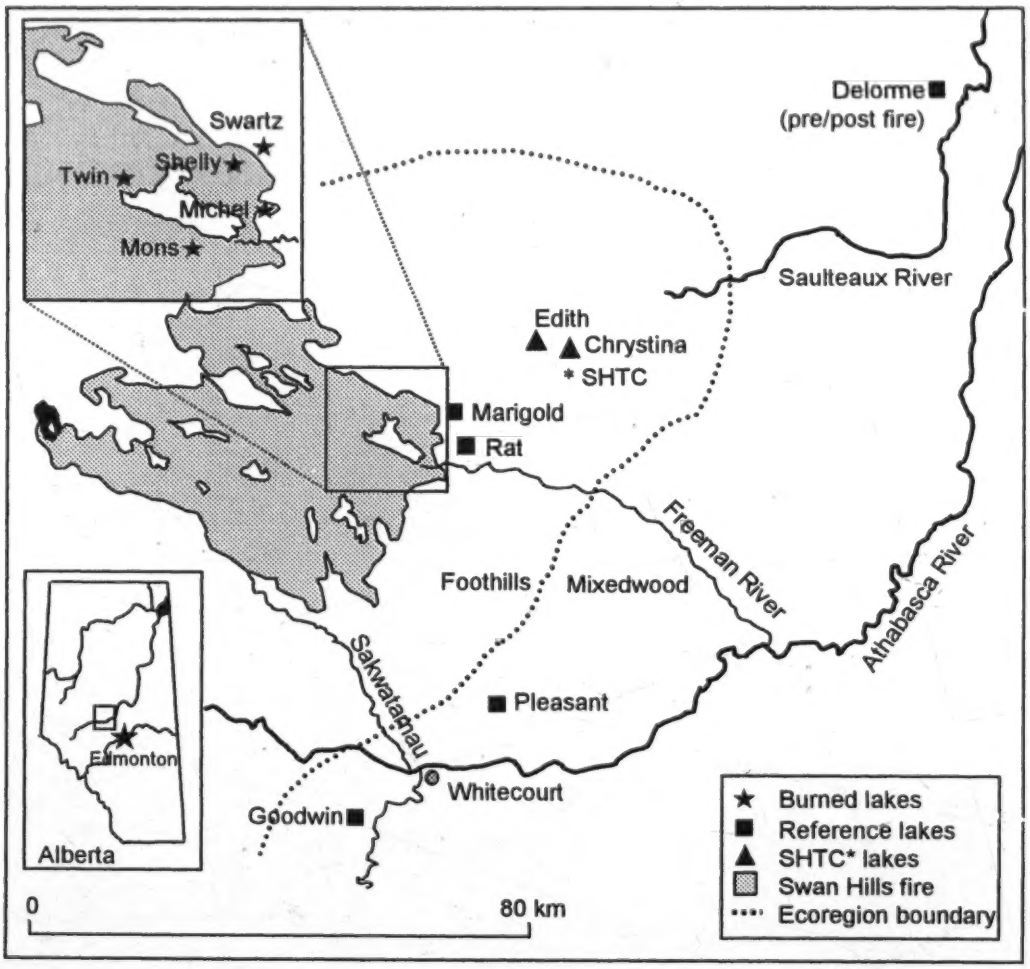

Figure 2-1. Swan Hills study lakes (*SHTC, Swan Hills Treatment Centre).

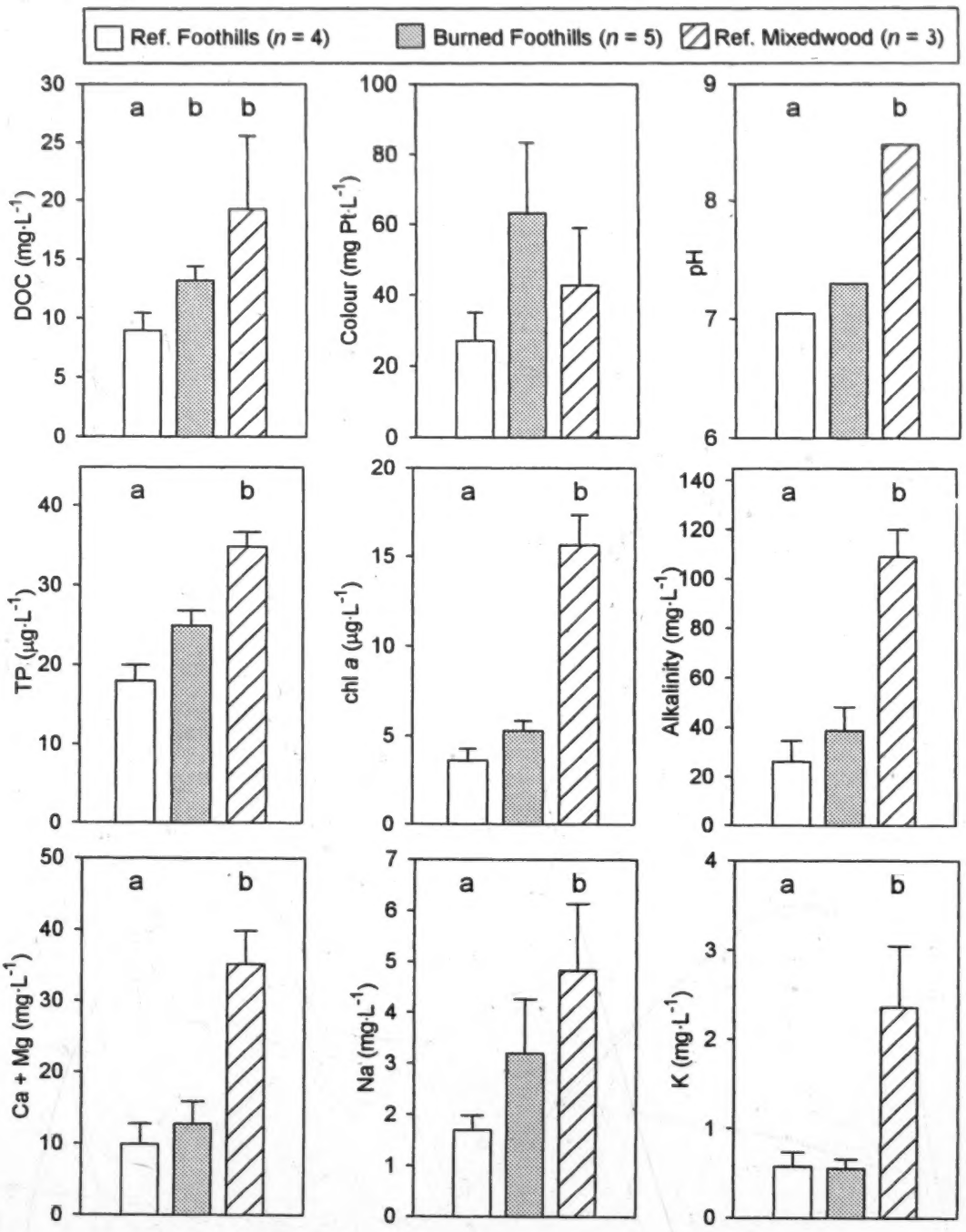

Figure 2-2. July to August means for water quality of reference Foothills, burned Foothills, and reference Mixedwood lakes (letters indicate differences between burned and reference Foothills lakes, and between reference Foothills and reference Mixedwood lakes; Mann-Whitney *U*-test; $P \leq 0.05$; whiskers represent one standard error of the mean; DOC, dissolved organic carbon; TP, total phosphorus; chl *a*; chlorophyll *a*).

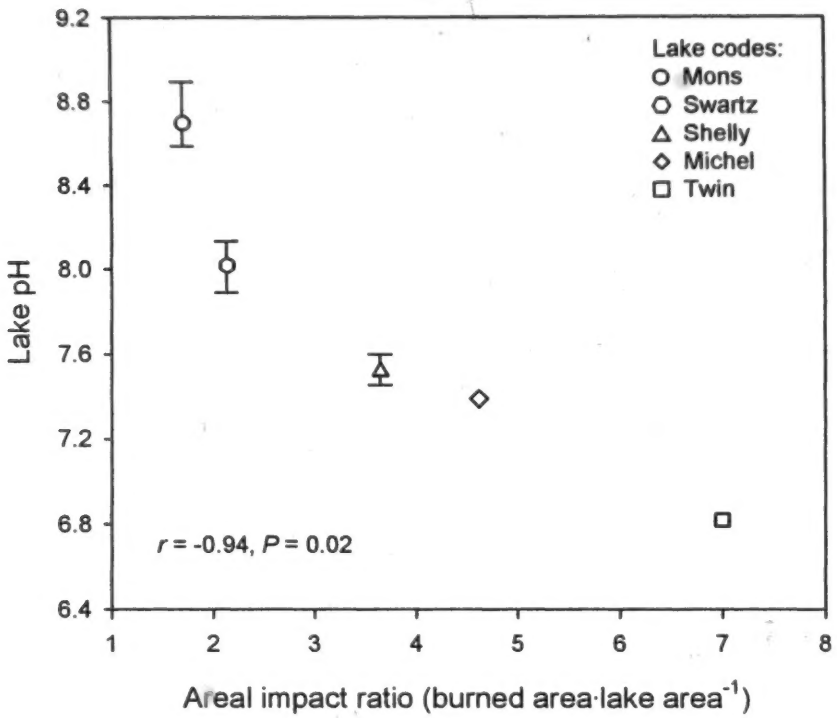

Figure 2-3. Scatterplot of lake pH and areal impact ratio from burned watersheds (bars represent the pH range from July to Aug. 2000).

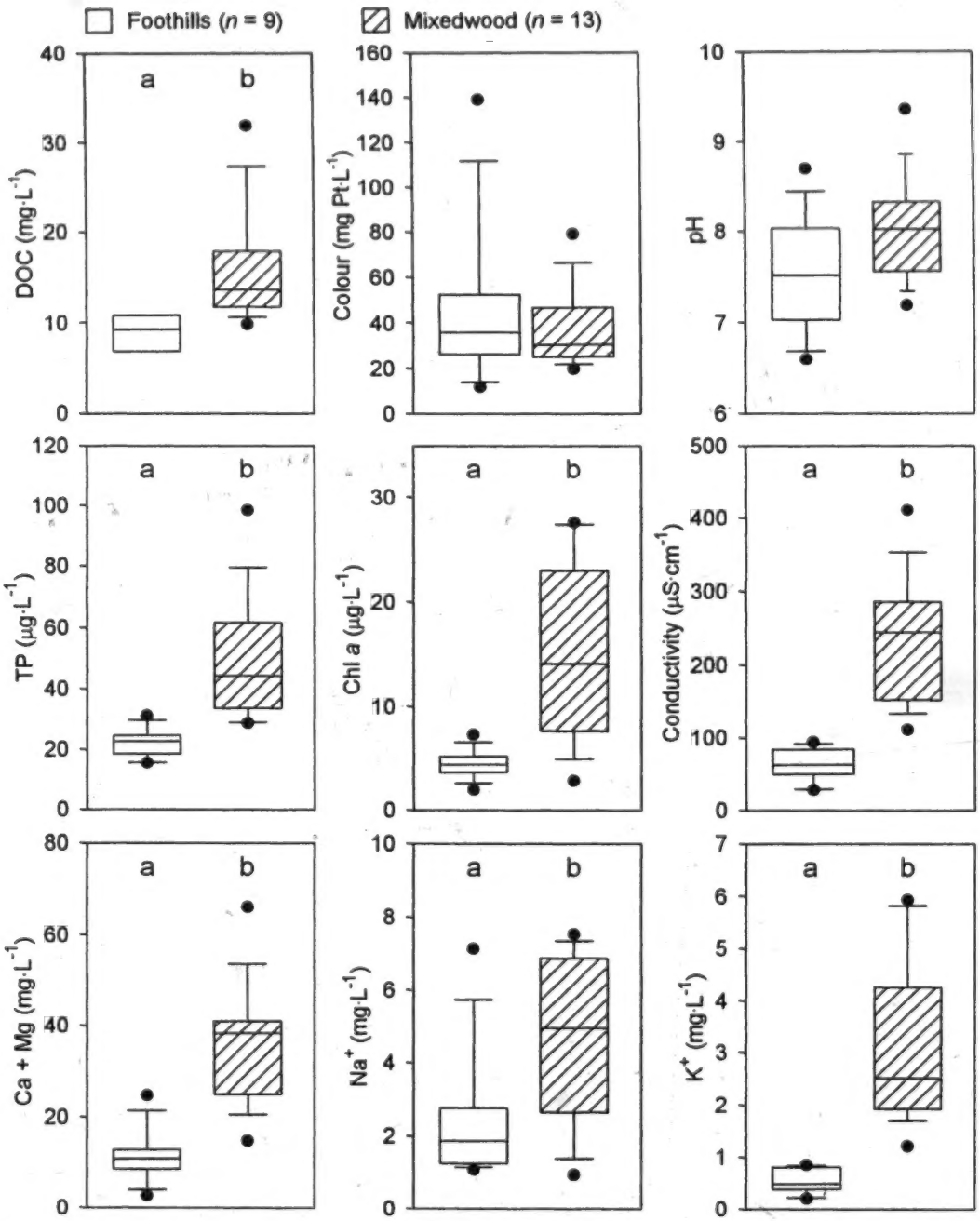

Figure 2-4. Boxplots of July to August means for water quality of Boreal Foothills and Boreal Mixedwood lakes (boxplots indicate medians, 10th, 25th, 75th, and 90th percentiles, and outliers; letters indicate differences between groups (*t*-test, $P < 0.05$; *$n = 4$ for Foothills lakes DOC; DOC, dissolved organic carbon; TP, total phosphorus; chl *a*; chlorophyll *a*).

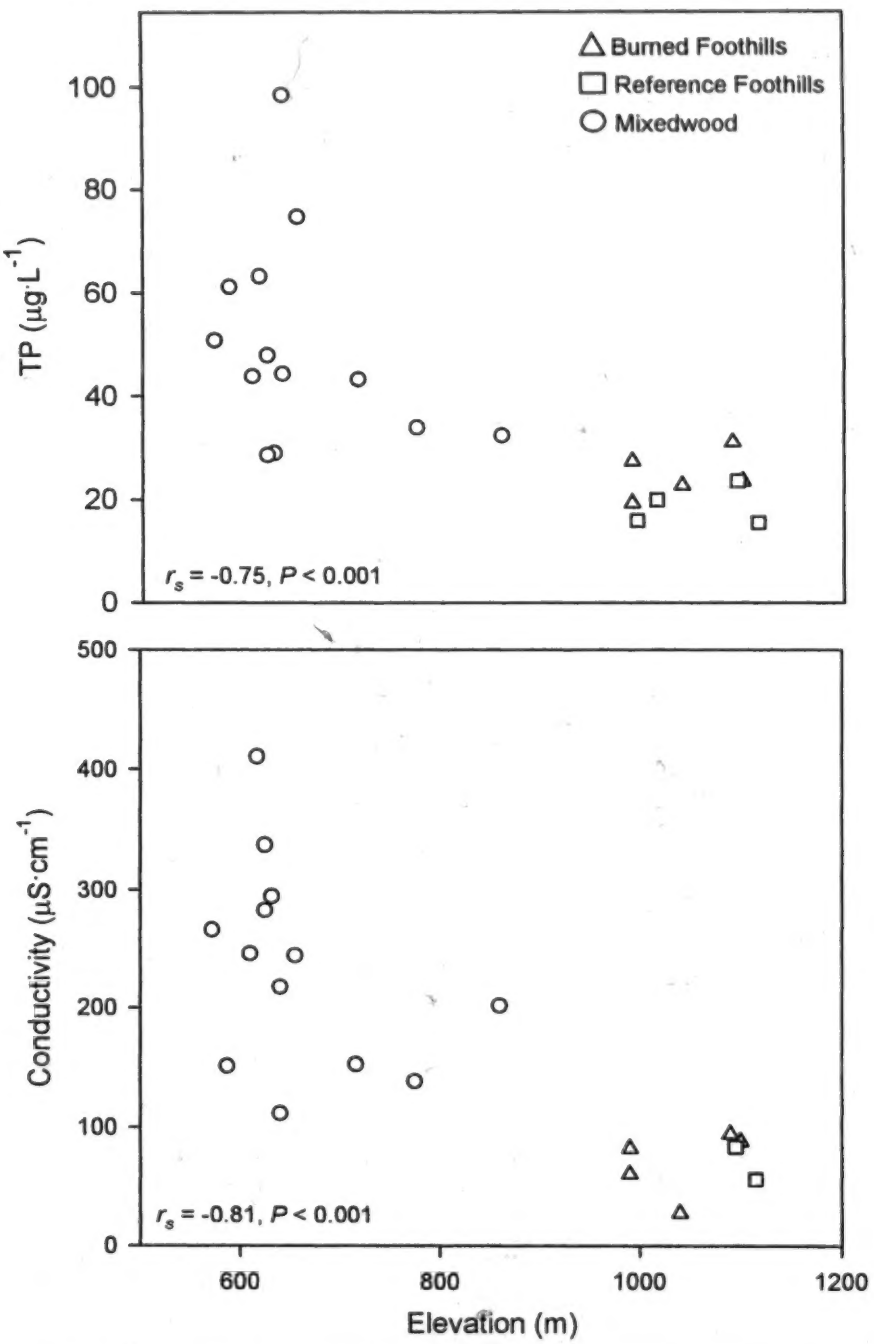

Figure 2-5. Scatterplots of total phosphorus (TP) concentration and conductivity vs. elevation for Boreal Foothills and Boreal Mixedwood lakes ($n = 22$; r_s, Spearman's rank correlation coefficient).

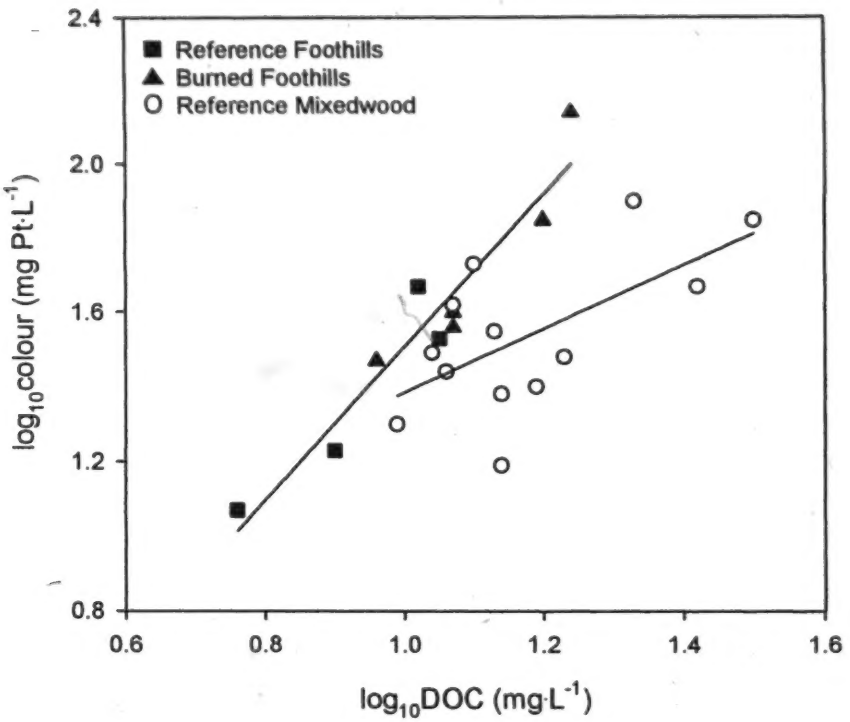

Figure 2-6. Relationship between lake colour and dissolved organic carbon (DOC) concentrations for Foothills ($r^2 = 0.92$, $P << 0.01$) and Mixedwood lakes ($r^2 = 0.40$, $P = 0.02$).

Chapter 3. INFLUENCE OF FOREST FIRE ON METHYL MERCURY IN MACROINVERTEBATE AND FISH COMMUNITIES ON THE BOREAL PLAIN

3.1 Introduction

Forest fire is the dominant natural disturbance in the boreal forest (Canadian Council of Forest Ministers 1997). As a sustainable disturbance, forest fire can provide a context for the ecological effects of forest cover removal on aquatic ecosystems (Carignan and Steedman 2000). Rapid expansion of the timber harvesting industry on Alberta's Boreal Plain has raised concerns over the impacts of forest disturbance on lakes and streams. Previously, these effects were thought to be limited to decreased water quality, however recent studies suggest that clear-cutting can increase mercury (Hg) concentrations in lake food webs (Garcia and Carignan 1999, 2000).

Hg is a pollutant of concern in aquatic ecosystems due to its toxicity and high rate of bioaccumulation (World Health Organization 1991; Becker and Bigham 1995). Aquatic biota accumulate Hg as methyl mercury (MeHg), a highly toxic compound produced by bacteria in lake sediments (Jensen and Jernelov 1969). The principal source of Hg to remote lakes is atmospheric deposition (Swain et al. 1992; Rudd 1995). Hg emissions from fossil fuel combustion, waste incineration, and other sources have increased the rate of Hg deposition in boreal watersheds (Louchouarn et al. 1993). As a consequence, many lakes in the boreal forest contain fish with Hg concentrations that exceed the Canadian consumption advisory limit of 0.5 $mg \cdot kg^{-1}$ (e.g., Wren et al. 1991; Garcia and Carignan 2000).

Watershed disturbance may affect MeHg bioaccumulation in several ways. Hg deposited in boreal soils is mobilized by the export of dissolved organic carbon (DOC; Lee and Iverfeldt 1991; Mierle and Ingram 1991), a process promoted by logging (Carignan et al. 2000) and forest fire (McEachern et al. 2000). Post-disturbance changes in lake chemistry may increase Hg methylation (i.e., pH; Miskimmin et al. 1992) and reduce MeHg photodegradation (i.e., colour; Sellers et al. 1996). Alternatively, a post-disturbance increase in primary productivity may reduce MeHg concentrations through biodilution (Larsson et al. 1992; Kidd et al. 1999). On the Boreal Shield, northern pike (*Esox lucius*) in lakes with logged watersheds had higher Hg concentrations and higher

lake water concentrations of DOC (2 yr post-disturbance) than burned and undisturbed watersheds (Garcia and Carignan 2000). These results suggest a greater impact from logging than from forest fire on MeHg, however it remains unclear whether forest disturbance has a similar effect on MeHg in Boreal Plain lakes.

Few studies have addressed MeHg bioaccumulation in Boreal Plain lakes. Unlike most boreal regions, the Boreal Plain is characterized by sedimentary geology, nutrient-rich soils, and low annual runoff. The trophic status of Boreal Plain lakes ranges from meso-oligotrophic to hyper-eutrophic (Mitchell and Prepas 1990), thus MeHg concentrations may be highly variable among lakes. Food webs in Boreal Plain lakes are dominated by littoral macroinvertebrates and small fish species such as brook stickleback (*Culaea inconstans*) (Beaudoin et al. 2001). Despite the importance of littoral food webs in the transfer of MeHg from the base of the food web to fish, previous research has focused primarily on MeHg uptake in pelagic systems (e.g., Plourde et al. 1997). Earlier studies indicate that MeHg bioaccumulation and biomagnification are controlled by a complex interaction of factors including water chemistry (Grieb et al. 1990), watershed morphometry (Driscoll et al. 1995), and trophic ecology (Cabana et al. 1994). It is not known, however, whether previously determined relationships between MeHg and environmental factors extrapolate to littoral food webs on the Boreal Plain.

The objective of the present study is to compare MeHg concentrations in littoral macroinvertebrates and fish from burned and unburned lakes on Alberta's Boreal Plain. As a further objective I examined the relationships between MeHg concentration, lake water chemistry, and trophic position among macroinvertebrates of various feeding strategies (e.g., herbivore, detritivore, and predator), and fish. Stable isotope ratios of nitrogen ($\delta^{15}N$) and carbon ($\delta^{13}C$) provided time-integrated estimates of food source ($\delta^{13}C$) and trophic position ($\delta^{15}N$; Peterson and Fry 1987).

3.2 Materials and Methods

3.2.1 Study area

In May 1998 a forest fire burned > 150 000 ha of boreal forest in the Swan Hills, including the watersheds of several streams and lakes (Fig. 3-1; for a full description of the study area, see Materials and Methods, Chapter 2). Lakes from five burned, headwater watersheds (mean disturbance±SE = 62±12%) and five undisturbed watersheds were selected based on lake size, wetland cover, and drainage ratio. Two additional lakes (Edith and Chrystina) were selected from unburned watersheds located within 10 km of the Swan Hills Treatment Center (SHTC), a hazardous waste incineration facility. Both SHTC lakes have fish consumption advisories due to high Hg concentrations in stocked brook trout (*Salvelinus fontinalis*) populations. Biota were collected and analyzed for MeHg to provide a relative benchmark for burned and reference lakes, which were remote from the SHTC.

Following sample collection in the 2000 open-water season (see below), half an undisturbed watershed (Delorme Lake) was logged as part of a prescribed timber harvesting experiment. The following spring, a forest fire burned the entire watershed, and the lake was resampled during the 2001 open-water season for a pre- vs. post-disturbance comparison of MeHg concentrations in biota and water chemistry. Thus I examined the influence of fire on MeHg bioaccumulation over two timeframes: 1) a pre- vs. post-disturbance comparison (3 months post-fire) in Delorme Lake, and 2) a burned vs. unburned comparison, 2 yr post-fire (1998 Swan Hills fire).

3.2.2 Lake sampling and analysis

For a description of lake sampling methods, water chemistry analyses, and watershed morphometry calculations, see Materials and Methods, Chapter 2. Four euphotic zone water samples were collected from Delorme Lake during the period of August to October 2000. Post-fire water samples from Delorme Lake were collected bi-weekly from August to October 2001. Summer rainfall (May to August) monitored at Slave Lake (30 km north-west of Delorme Lake) was 7% higher in 2001 (460 mm) than in 2000 (430 mm).

Water samples for total Hg analysis (THg) were collected from 8 lakes (4 burned, 4 reference) in September 2000. Duplicate surface samples were collected from the middle of each lake with Teflon© jars. The jars were filled with 0.2% BrCl and HCl rinse solution, and were emptied shortly before sample collection. Polyethylene gloves were worn to minimize sample contamination. During sample collection, one person handled only the jar while another person handled the Ziplock bags used for storage. An HCl preservative was added to each jar after sample collection.

Aquatic macroinvertebrates were collected from all 12 lakes in September 2000 and 2 lakes (Delorme and Goodwin) in late August 2001. Goodwin Lake watershed was undisturbed between 2000 and 2001 and served as a reference to Delorme Lake. The following macroinvertebrates encountered in the lakes and were collected where they occurred: *Agrypnia* spp. and *Phryganea* spp. (Trichoptera: Phryganeidae), *Gammarus lacustris* (Crustacea: Amphipoda), *Aeshna* spp. (Odonata: Aeshnidae), *Cordulia shurtleffi* (Odonata: Corduliidae), and *Notonecta* spp. (Hemiptera: Notonectidae) (Table 3-1). The two genera of Trichoptera (*Agrypnia* spp. and *Physa* spp.) appeared to be mutually exclusive in the lakes and were treated as a single taxon. The grazer *Physa* spp. (Gastropoda: Physidae) was collected to adjust for baseline variation in $\delta^{15}N$ (Post 2002), but was not analyzed for MeHg.

Macroinvertebrates were collected from sampling stations randomly selected in the littoral zones of the study lakes. At each station, macrophytes and surficial sediments were swept several times with a pond net (mesh size: 1000 µm). After each sweep, all visible macroinvertebrates were picked with Teflon©-coated forceps and placed in plastic buckets filled with lake water. Within 2 h of sample collection, macroinvertebrates were live-sorted to Family or Genus and placed in separate polyethylene pans filled with deionized distilled water (DDW) for 12 to 15 h to allow for defecation of gut contents. Macroinvertebrates were then blotted for ~1 min on Whatman 12.5 cm filter paper, sealed in sterilized polyethylene Whirl-Pak bags and frozen on dry ice (-20°C). All biota were keyed out to Subfamily, Genus, or species with Clifford (1991) and Nelson and Paetz (1992).

Brook stickleback (*Culaea inconstans*) occurred in 2 burned (Shelly and Michel), 4 reference (Goodwin, Pleasant, Marigold, and Delorme), and both SHTC lakes (Edith

and Chrystina). Pearl dace (*Margariscus margarita*) occurred in Delorme Lake only. Fish were collected with 6 to 8 Gee minnow traps set along 200 m of shoreline. Minnow traps were set 1 to 3 m from shore at depths of 0.5 to 1 m, and set times ranged from 4 to 5 h. Baitfish were transferred to plastic buckets of lake water, live sorted within 1 hr of sampling, frozen on dry ice and sealed in Whirl-Pak© bags.

3.2.3 MeHg and THg analyses

Macroinvertebrates were wet weighed (Mettler AT261 DeltaRange) and body lengths measured prior to freeze-drying for 70 h at –60°C and 100 millitorr using a Freezemobile 24 (Virtis Model 6201-2220). With the exception of Odonates (*Aeshna* spp. and *C. shurtleffi*), aggregate samples of 5 to 15 individuals were needed to meet the required mass for MeHg and stable isotope analyses (50 to 100 mg). Odonates were large enough to be analyzed as individuals. Dried samples were homogenized in Whirl-Pak bags by rolling a glass pestle over the outside of the bag. Between 50 and 100 mg of each homogenate were transferred to glass vials (Qorpak 14.8 ml w/Teflon lined closures) and shipped to Flett Research Ltd. (Winnipeg, MB) for MeHg analysis. Where possible, 10 replicates of each taxon were analyzed per lake (mean replicate size = 9±2; range = 2 to 12).

C. inconstans and *M. margarita* were measured for total length, weighed, sexed, eviscerated, and grouped into aggregate samples of 3 to 5 male fish. To determine if MeHg concentrations varied with body size, I grouped the fish according to size and formed homogenates that represented the size range in each lake. The fish were freeze-dried prior to homogenization in a stainless steel blender cup (12 to 37 ml capacity) using a Waring Model 700G blender. Between 200 to 400 mg of homogenate were transferred to glass vials and shipped to Flett Research Ltd.

To prevent contamination, all glassware and polyethylene containers used in sample preparation were soaked in 10% HNO_3 (TraceMetals Grade) and thoroughly rinsed with distilled, deionized water. Polyethylene gloves were worn throughout sample preparation. Because MeHg is concentrated in tissue relative to the environment, the risk of contamination was considered to be low.

MeHg in tissue was detected with cold vapour atomic fluorescence spectrophotometry (CVAFS) at Flett Research Ltd., Winnipeg, MB. Detection of MeHg

followed overnight digestion in KOH methanol, ethylation with tetraethyl borate, collection on Tenax traps, and separation on a gas chromatograph (Horvat et al. 1993). Average coefficient of variation for duplicate analyses (2 subsamples from 1 digestion), performed on 10% of the samples (n = 74), was 8.1±0.7%. Repeat analyses (2 digestions prepared) were also performed on 10% of the samples (n = 80) with an average coefficient of variation of 7.1±0.6%. Standard materials (DORM-2; MeHg concentration = 4.47±0.34 $ng \cdot g^{-1}$) were run every ~8 samples to determine % recovery (mean recovery = 86±0.5%; n = 135). To determine the recovery of labile MeHg, spiked tissue samples were analyzed every 8 samples (mean recovery = 96±0.7%, n = 132).

Lake water was analzyed for THg with CVAFS at the Canadian Centre for Inland Waters (CCIW), Burlington, ON. Detection of Hg followed digestion in 0.1 N bromine monochloride. Hg concentrations were determined against standard concentration curves generated for each daily run and were accompanied by a suite of QA/QC samples including reagent blanks, spikes, and two different dilute standard reference waters. Samples analyzed in triplicate had a mean coefficient of variation of 6.3%.

3.2.4 Stable isotope analysis

Stable isotope ratios of nitrogen (^{15}N:^{14}N; $\delta^{15}N$) and carbon (^{13}C:^{12}C; $\delta^{13}C$) were used to examine the relationship between trophic position and MeHg, and to compare trophic positions and food web structure among burned and reference lakes. Stable isotope ratios are commonly used to define food web structure and trace trophic interactions in aquatic and terrestrial ecosystems (e.g., Peterson and Fry 1987). Trophic level in consumers can be estimated from $\delta^{15}N$ because ^{14}N is excreted preferentially to ^{15}N, thus the $\delta^{15}N$ signal in tissue becomes enriched with each succeeding trophic level. The average fractionation between trophic levels is approximately 3.4‰ when averaged over multiple trophic pathways (Vander Zanden and Rasmussen 1999; Post 2002). To adjust for baseline variation in $\delta^{15}N$, gastropod $\delta^{15}N$ values were subtracted from consumer $\delta^{15}N$ ($\delta^{15}N_{adj} = \delta^{15}N_{consumer} - \delta^{15}N_{snail}$; Post 2002). Trophic position (TPOS) was calculated as follows: $TPOS = (\delta^{15}N_{adj}/3.4) + 2$ (Post 2002), where gastropods are assumed to occupy the second trophic level.

Carbon isotope ratios do not fractionate to a large extent through trophic interactions, and reflect variation in carbon sources at the base of food chains. Earlier

studies indicate that profundal, pelagic, and littoral carbon sources can be distinguished with $\delta^{13}C$ (e.g., Vander Zanden and Rasmussen 1999), and this technique has been applied in previous studies to compare the diets of consumers in lakes (Rau 1980). The ratio of C:N is also a useful tool, and can be used to compare lipid content among taxa (e.g., Anderson 1994).

Stable isotope ratios were calculated as the parts per mil (‰) deviation from standard material (i.e., $\delta^{15}N = ([R_{sample}/R_{standard}]-1) \times 1000$, where $R = {}^{15}N/{}^{14}N$). Atmospheric nitrogen (N_2) was used as the reference material for $\delta^{15}N$ and Pee Dee belemnite (PDB) limestone was used for $\delta^{13}C$. The $R_{standard}$ values of each reference material were set at 0‰. Macroinvertebrate homogenates prepared for MeHg analysis were sub-sampled for stable isotope analysis. For all tissues, 1.2 to 1.7 mg of material were weighed on a Cahn C31 Microbalance and packed into 8 x 5 mm tin capsules, which were then folded into a square. Snail foot muscle tissue was dissected from the shell, freeze-dried, and homogenized with a glass mortar and pestle. Dorsal muscle tissue was dissected from each of the fish in the aggregate samples prepared for MeHg analysis prior to homogenization. The muscle samples were freeze-dried for 50 h at –60°C, homogenized, and packed in tin capsules. Stable isotope ratios were measured with a Micromass Optima continuous flow mass spectrometer (CF-IRMS) coupled to a Carlo Erba Na1500 elemental analyzer (EA) and Autosampler at the National Water Research Institute (Saskatoon, SK). Standards were analyzed every 10 samples; average coefficients of variation for duplicate samples ($n = 14$) were 2.1±0.6% for $\delta^{13}C$ and 9.6±2.0% for $\delta^{15}N$.

3.2.5 Statistical analysis

Pre- and post-disturbance MeHg concentrations, stable isotope ratios, and body size data from Delorme Lake were compared with Student's *t*-test. Due to low sample size, the Mann-Whitney *U*-test was used to compare water chemistry in Delorme Lake between pre- and post-disturbance samples ($n = 4$ and 8, respectively). MeHg, water chemistry, stable isotope ratios, and body size data from lakes burned in the 1998 Swan Hills fire were compared to unburned lakes with the Mann-Whitney *U*-test. Because all the burned lakes were located in the Boreal Foothills ecoregion, I used the Mann-Whitney test in two-way comparisons between burned and reference Foothills lakes, and

between reference Foothills and reference Mixedwood lakes. Lake means for C:N ratios were compared among taxa with ANOVA. Pearson correlation coefficients were used to examine the influence of environmental variables such as water chemistry and landscape features on MeHg bioaccumulation and biomagnification. Relationships between $\delta^{15}N$ signatures and MeHg concentrations were examined with simple linear regression. Biomagnification rates were calculated from the linear regression of $\delta^{15}N$ and MeHg concentrations. Biomagnification factors between trophic levels were calculated as the fraction of MeHg concentration between upper and lower trophic levels. All variables were tested for normality using the Shapiro-Wilkes test. Where necessary, non-normal variables were $\log_{10}$ transformed prior to analysis of variance. Proportions were arcsine transformed. Variables that were not normalized by transformation were analyzed with non-parametric methods. All statistical analyses were conducted with SPSS (v. 8.0) and results were considered significant at $P \leq 0.05$.

3.3 Results

3.3.1 Lake water chemistry

Delorme Lake Fire (2001)

Several changes in the water chemistry of Delorme Lake were detected in the summer following watershed disturbance. Post-fire concentrations of NH_4^+ and $NO_2^- + NO_3^-$ in lake water increased by 12-fold (Mann-Whitney *U*-test, $P < 0.01$) and 3.8-fold ($P < 0.01$) respectively, suggesting a substantial post-fire release of inorganic N (Fig. 3-2). There were no detectable changes in concentrations of lake water TP ($P = 0.2$) or chl *a* ($P = 0.5$) between pre- and post-fire years. Lake water DOC concentration increased by 1.2-fold ($P < 0.01$), however there was no significant change in colour ($P = 0.7$). Alkalinity increased by 1.2-fold after the fire ($P < 0.01$). Elevated DOC and inorganic nitrogen concentrations in lake water are consistent with short-term effects of forest fire reported in earlier studies (e.g., McEachern et al. 2000).

Swan Hills Fire (1998)

Lake water chemistry in the Swan Hills appeared to vary over an elevation gradient (i.e., 610 to 1115 m), becoming progressively dilute with increasing elevation (Table 3-2). Three reference lakes located in Mixedwood catchments at the base of the

Swan Hills (Goodwin, Delorme and Pleasant) had higher lake water concentrations of TP, chl *a*, DOC, and dominant cations than reference lakes in conifer-dominated catchments located at higher elevations. The differences in water chemistry between unburned Mixedwood and Foothills watersheds are likely related to ecoregional factors such as vegetation, landscape position, and local differences in climate. Since the burned lakes were all located in conifer-dominated watersheds in the Boreal Foothills, their water chemistry was not comparable to that of reference Mixedwood watersheds.

Compared to unburned lakes in the Foothills ecoregion ($n = 4$), burned Foothills lakes ($n = 5$) had 1.4-fold higher lake water DOC concentration (Mann Whitney *U*-test, $P = 0.05$) and showed similar (but undetectable) increases in concentrations of TP (1.3-fold, $P = 0.1$) and chl *a* (1.5-fold, $P = 0.08$). Inorganic N concentrations were not different between burned and unburned Foothills lakes ($P = 0.3$). Mean THg concentrations in surface water samples of burned lakes was 1.4±0.2 $ng \cdot L^{-1}$, and were not significantly different from reference lakes (2.7±1.5 $ng \cdot L^{-1}$, $P = 1$). Among burned lakes, colour increased with the ratio of area burned to lake volume (volumetric impact ratio, $r = 0.95$, $P = 0.01$) and lake water pH decreased with the ratio of area burned to lake area (areal impact ratio, $r = -0.94$, $P = 0.02$). These data appear to indicate a modest, short-term increase in the export of terrestrial humic substances to burned lakes.

3.3.2 Stable isotope ratios

Delorme Lake

There were modest differences between stable isotope ratios ($\delta^{15}N$ and $\delta^{13}C$) and body sizes of the biota collected from Delorme Lake between pre- and post-disturbance years (Table 3-3). $\delta^{15}N$ signatures in *Phryganea* spp. and *G. lacustris* increased slightly from 2000 to 2001 (+0.9‰ and 0.5‰ respectively, *t*-test, $P << 0.01$). Increases in the $\delta^{15}N$ of primary consumers may reflect post-fire input of dissolved inorganic nitrogen (e.g., Lake et al. 2001). Mean body lengths of *Phryganea* spp. and *G. lacustris* were 6 and 13% larger in 2001 than in 2000 ($P = 0.03$ and $<< 0.01$, respectively). Pre- and post-disturbance $\delta^{13}C$ did not change in *Phryganea* spp. or *G. lacustris* ($P = 0.3$ and 0.7, respectively). *C. shurtleffi* $\delta^{15}N$ and $\delta^{13}C$ were depleted by 0.3‰ and enriched by 1.3‰, respectively ($P = 0.05$). There were no differences in body length, $\delta^{15}N$, and $\delta^{13}N$ in

Notonecta spp., *M. margarita,* and *C. inconstans* homogenates collected in pre- and post-disturbance years.

Swan Hills lakes

Baseline variation in stable isotope ratios ($\delta^{15}N$ and $\delta^{13}C$) appeared to reflect physico-chemical properties of the study lakes. Gastropod (*Physa* spp.) $\delta^{15}N$ values ranged from 1.0 to 4.3‰ and were strongly correlated with lake area ($r = 0.74$, $P = 0.01$), as observed by Post (2002). Gastropod $\delta^{13}C$ signatures varied from –16.3 to –29.5‰, possibly reflecting the broad range in lake water pH (Osmond et al. 1981) and variation among sources of dissolved inorganic carbon (DIC) to lakes (Post 2002).

There were no significant differences in baseline-adjusted $\delta^{15}N$ ($\delta^{15}N_{adj}$) between burned ($n = 5$) and unburned lakes ($n = 7$, Mann Whitney, $P > 0.07$). *Aeshna* spp. and *C. inconstans* in SHTC lakes were trophically elevated by 1.4‰ and 3.1‰ relative to the mean from burned and reference lakes ($P = 0.04$ and 0.05, respectively). These differences may reflect variation in food web structure between the meso-oligotrophic SHTC lakes, and mesotrophic lakes at lower elevations.

Stable isotope analysis indicated a trophic gradient spanning 3 levels from primary consumer (Trichoptera) to top predator (*C. inconstans*). Mean trophic positions estimated from $\delta^{15}N_{adj}$ of macroinvertebrates spanned only one trophic level (3.4‰ $\delta^{15}N$), increasing through Trichoptera, *G. lacustris*, *C. shurtleffi*, *Notonecta* spp., and *Aeshna* spp. (Fig. 3-3, Table 3-4). $\delta^{15}N$ values among macroinvertebrates were comparable those reported for similar taxa on the Boreal Shield (Kidd et al. 1999). Brook stickleback (*C. inconstans*) were elevated by one trophic level over the top macroinvertebrate predator, *Aeshna* spp. The trophic position of Trichoptera (TPOS, range = 0.7 to 2.8) appeared to increase from alkaline to acidic lakes, as indicated by the inverse correlation between baseline-adjusted $\delta^{15}N$ and lake pH ($r = 0.78$, $P < 0.01$). Trichoptera $\delta^{13}C$ signatures (mean = –20.5±1.3‰) were generally enriched relative to other taxa (Fig. 3-3), suggesting a diet of macrophytes (Lazerte 1983). In lakes where Trichoptera were trophically elevated, their $\delta^{13}C$ signatures were less distinct from other taxa, suggesting a shift in diet across lakes.

G. lacustris, a detritivore, was trophically elevated relative to Trichoptera, and appeared to supplement the diet of the bottom-dwelling predator *C. shurtleffi* (Merrit and Cummins 1996). *G. lacustris* and *C. shurtleffi* shared similar $\delta^{13}C$ signatures in 3 of 4 lakes they co-inhabited (-25.2±1.2 and –25.5±0.9‰, respectively), however their $\delta^{15}N$ signatures were separated by only 1.0±0.2‰ (Mann Whitney *U*-test, $P = 0.05$), suggesting omnivory by one or both taxa. Among lakes, trophic position varied from 2.1 to 2.8 in *Aeshna* spp., 2.2 to 2.9 in *Notonecta* spp., and 3.4 to 5.1 in *C. inconstans*. C:N ratios were highest in *Notonecta* spp. (5.5±0.1) and lowest in *C. inconstans* (3.4±0.01; ANOVA, $P << 0.01$). Mean $\delta^{13}C$ signatures of *Notonecta* spp., *Aeshna* spp., and *C. inconstans* were 25.7±1.1, -24.8±0.6, and -24.0±1.1‰. Based on an earlier study from the Boreal Plain, these $\delta^{13}C$ values reflect a littoral food source (Beaudoin et al. 2001). Together the stable isotope data indicate different carbon sources among littoral consumers, a high degree of omnivory, and within-taxon variation of trophic position among lakes.

3.3.3 MeHg in aquatic biota

Delorme Lake

MeHg concentrations in macroinvertebrates and fish from Delorme Lake decreased after the fire (Fig. 3-4). Compared to pre-fire values, MeHg concentrations decreased by 2-fold in *Phryganea* spp. (*t*-test, $P << 0.01$), 1.8-fold in *G. lacustris* ($P << 0.01$), 1.5-fold in *C. shurtleffi* ($P = 0.03$), 1.6-fold in *M. margarita* ($P < 0.01$) and 1.7-fold in *C. inconstans* ($P << 0.01$, Fig. 3-4). Over the same time period (2000 to 2001), MeHg concentrations in similar taxa from a reference lake (Goodwin) did not change (*G. lacustris* ($P = 0.4$), *C. shurtleffi* ($P = 0.2$), and *C. inconstans* ($P = 0.3$)). As indicated by the stable isotope ratios, there were no major changes in trophic position among the biota collected from pre- and post-fire Delorme Lake, therefore the decrease in MeHg concentrations in biota may be related to post-fire fluctuations in water chemistry.

Swan Hills Lakes

With the exception of *C. shurtleffi*, MeHg concentrations in biota were not significantly different between burned and reference lakes in the Swan Hills (Mixedwood and Foothills lakes combined; Fig. 3-5). Mean MeHg concentration in *C. shurtleffi* was 1.4-fold higher in burned lakes (55±4 $ng \cdot g^{-1}$) relative to reference lakes (38±6 $ng \cdot g^{-1}$, $P =$

0.05). This result was likely attributable to the mitigating effect of alkaline, highly productive conditions on MeHg bioaccumulation in Mixedwood lakes, not the influence of forest fire. When the burned vs. unburned comparison was restricted to the Foothills ecoregion, there was a general trend of lower MeHg concentrations in biota from burned lakes compared to unburned lakes (Fig. 3-6), however none of the differences were significant.

MeHg concentrations in macroinvertebrates and fish were correlated with lake chemistry and watershed characteristics. When data from all lakes were pooled, MeHg concentrations in Trichoptera and *G. lacustris* were positively correlated with conifer cover, which dominated the Foothills catchments (r = 0.73 and 0.86, P = 0.02 and 0.01, respectively). Conifer cover was inversely correlated with lake water TP concentration (r = -0.65, P = 0.02) and pH (r = -0.62, P = 0.03), thus correlations with MeHg concentrations in Trichoptera and *G. lacustris* appeared to reflect differences in lake chemistry between the ecoregions. MeHg concentrations in *C. shurtleffi*, *Aeshna* spp., and *C. inconstans* were all inversely, and similarly, correlated to lake water pH (Fig. 3-7) and TP concentrations (r = -0.80, -0.84, and –0.78, P = 0.02, <0.01, and 0.02, respectively). MeHg concentrations in *Aeshna* spp. were also correlated with percent wetland cover (r = 0.87, P < 0.01) and lake water concentrations of chl *a* (r = -0.73, P = 0.03). MeHg concentrations in *Notonecta* spp. were inversely correlated with lake water TDP concentration (r = -0.65, P = 0.02). Together these correlations suggest that MeHg concentrations in the lakes were influenced by an elevational gradient in lake productivity.

The relationships between MeHg concentrations in biota and lake water pH (Fig. 3-7) suggest that MeHg bioaccumulation in lakes near the Swan Hills Treatment Centre (SHTC), though generally higher than the other lakes, was not higher than expected based on water chemistry. Both SHTC lakes (Edith and Chrystina) were among the lowest in pH and TP concentration among the lakes (Table 3-2). Further, many of the correlations presented in Table 3-5 remain significant when the SHTC lakes are removed, indicating that inclusion of these lakes has not skewed the relationships between lake chemistry and MeHg concentrations in biota.

3.3.4 Biomagnification of MeHg

Mean MeHg concentrations ranged among macroinvertebrate taxa from 16±1 $ng \cdot g^{-1}$ in Trichoptera (n = 10), to 29±2 $ng \cdot g^{-1}$ in *G. lacustris* (n = 7), 57±8 $ng \cdot g^{-1}$ in *C. shurtleffi* (n = 8), 102±21 $ng \cdot g^{-1}$ in *Aeshna* spp. (n = 9), and 181±10 $ng \cdot g^{-1}$ in *Notonecta* spp. (n = 12). Mean MeHg concentration in brook stickleback was 161±32 $ng \cdot g^{-1}$ (n = 10). Differences in MeHg concentrations between *G. lacustris* and *C. shurtleffi*, and between *C. inconstans* and *Notonecta* spp. were not significant (Mann Whitney U-test, P = 0.2 and 0.1 respectively). Over half (53%) of the variation in MeHg concentrations among taxa was explained by $\delta^{15}N_{adj}$ (Fig. 3-8, r^2 = 0.53, $\log_{10}MeHg = 1.7+0.13*\delta^{15}N_{adj}$, $P << 0.01$, n = 54). The unexplained variation in MeHg concentrations was due in part to lake effects, but also appeared to be attributable to taxon-specific factors. For example, *Notonecta* spp. appeared to accumulate MeHg more efficiently than other taxa (Fig. 3-8). Despite individual exceptions, the relationship between MeHg concentrations in biota and $\delta^{15}N_{adj}$ indicated biomagnification of MeHg in the littoral food web.

Among lakes, biomagnification factors (i.e., increase in MeHg concentration per trophic level) appeared to vary with lake chemistry. On average, MeHg concentrations in biota increased 10-fold over the 2 trophic levels that spanned Trichoptera and brook stickleback (*C. inconstans*). Among lakes, biomagnification factors between Trichoptera and brook stickleback ranged from 6 to 14 and were negatively correlated with chl *a* concentrations (r = -0.77, P = 0.04, n = 7). The trophic distance ($\delta^{15}N$) separating Trichoptera and brook stickleback was inconsistent, ranging from 5.3 to 9.9‰ (1.6 to 2.9 trophic levels) and unrelated to biomagnification factors (r = -0.36, P = 0.42). Because fish were absent in some lakes, I determined the rate of biomagnification of MeHg among macroinvertebrates only (i.e., between Trichoptera and *Notonecta* spp.). Significant, positive relationships were detected between $\log_{10}MeHg$ and $\delta^{15}N$ in 9 of the 10 lakes containing trichopterans, odonates, and *Notonecta* spp., with slopes ranging from 0.13 to 0.37. As with the biomagnification factor, the slopes of the $\log_{10}MeHg$-$\delta^{15}N$ relationships appeared to decrease with increasing lake productivity (e.g., TP concentration, r = -0.82, $P < 0.01$, Fig. 3-9). These data suggest that biomagnification of MeHg may be less efficient in lakes with high productivity, and may reflect the influence of growth biodilution on MeHg concentrations in biota.

Body size did not appear to be a confounding factor in among-lake comparisons of MeHg concentrations in *C. inconstans*. With the exception of Marigold Lake, within-lake correlations between MeHg concentration in *C. inconstans* and body size were non-significant. In Marigold Lake, MeHg concentrations in *C. inconstans* were negatively correlated with body size (r = -0.73, P= 0.04), contrary to positive correlations reported for other fish species (e.g., Grieb et al. 1990). MeHg concentrations in *M. margarita* from Delorme Lake increased with body length (r = 0.80, $P << 0.01$) and $\delta^{13}C$ (r = 0.73, $P << 0.01$), and may reflect a shift in prey selection between small and large-bodied fish.

3.4 Discussion

3.4.1 Influence of forest fire on lake chemistry and MeHg

The post-fire decline in MeHg concentrations in Delorme Lake biota may be linked to increases in lake water DOC concentration, productivity, and alkalinity. Unlike previous studies, the post-fire increase in DOC concentration in Delorme Lake water was not accompanied by increased colour (e.g., McEachern et al. 2000), suggesting in-lake production of organic carbon. Allochthonous inputs of organic carbon are typically accompanied by an increase in lake water colour due to the refractive properties of humic and fulvic acids (e.g., Rasmussen et al. 1989). Autochthonous carbon, comprised primarily of low molecular weight compounds produced by algae and macrophytes, do not affect the colour of lake water to the same extent as humic substances (Wetzel 2001). Laboratory experiments indicate that high concentrations of allochthonous DOC reduce the bioavailability of Hg to microorganisms (Barkay et al. 1997) and decrease Hg methylation rates (Miskimmin et al. 1992). The influence of autochthonous DOC on Hg biogeochemistry is less clearly defined, but most likely has a similar effect. Hg has a strong affinity for organic compounds with sulf-hydroxyl groups, amino acids, and hydroxy carboxylic acids (Neff 2002), and complexation by autochthonous organic matter may reduce the bioavailability of organic contaminants in productive lakes (e.g., Larsson et al. 1992).

Many studies report positive correlations between DOC concentrations in lake water and Hg concentrations in fish (e.g., Driscoll et al. 1994; McMurtry et al. 1989), however the majority of this research focuses on lakes with low concentrations of DOC

(i.e., < 10 $mg \cdot L^{-1}$; Carignan et al. 2000) comprised primarily of allochthonous substances. Allochthonous DOC inputs are typically associated with flux of Hg from the watershed, and low pH and high colour in lake water (Lee and Iverfeldt 1991; Rasmussen et al. 1989). These factors appear to promote net Hg methylation rates (Xun et al. 1987; Miskimmin et al. 1992; Sellers et al. 1996) and MeHg bioavailability (Miller and Akagi 1979), and thereby counteract the inhibitory effect of adsorption to allochthonous DOC. Because the production of autochthonous DOC in Delorme Lake was not associated with increased input of Hg, acidity, and colour, it likely had a net negative effect on MeHg concentrations.

Increased production of autochthonous DOC in Delorme Lake may have been stimulated by post-fire inputs of dissolved inorganic nitrogen (DIN). Elevated concentrations of DIN in lake water have previously been associated with forest fire (e.g., McEachern et al. 2000; Carignan et al. 2000) and are attributed to leaching of mineralized N from soils (Tiedemann et al. 1978). An increase in biomass or growth rates associated with elevated productivity could have further contributed to the reduction in MeHg concentrations through biodilution of the MeHg pool (Larsson et al. 1992; Garcia and Carignan 1999). Post-fire inputs of NH_4^+ and other base cations may have also contributed to the increase in alkalinity in Delorme Lake water. High alkalinity and water hardness appear to reduce the uptake of waterborne MeHg in aquatic biota (Rodgers and Beamish 1983), possibly through competitive inhibition by divalent cations (e.g., Ca^{2+}; Pagenkopf 1983). Decreased concentrations of bioavailable MeHg in the water column would translate into a reduction of MeHg concentrations in primary producers and affect a decrease throughout the entire food web, as was observed in Delorme Lake.

Two years post-fire, MeHg concentrations in burned Foothills lakes (1998 fire) were not detectably different from reference lakes, despite an apparent increase in allochthonous DOC inputs. Lake water colour, acidity, and light extinction increased in the burned lakes with the ratio of area burned to lake size, suggesting post-fire input of humic substances. However, post-fire inputs of allochthonous DOC and associated changes in lake chemistry may have been insufficient to increase MeHg concentrations in biota. Further, post-fire release of nutrients may have increased primary productivity in

the burned-Foothills lakes, and the apparent increase in DOC concentrations may have included some autochthonous production of organic carbon. Overall my results from Delorme Lake and the burned Foothills lakes agree with those of previous studies, and indicate that the impacts of forest fire on water chemistry either inhibit, or have no effect on the accumulation of MeHg in aquatic biota.

3.4.2 Influence of ecoregion on lake water chemistry and MeHg

Comparison of MeHg concentrations between burned and reference watersheds was complicated by variability in watershed features and lake water chemistry between Boreal Mixedwood and Boreal Foothills ecoregions. MeHg concentrations in littoral macroinvertebrates and brook stickleback were correlated with lake water pH, water hardness, and productivity indices, all of which decreased along an elevational gradient from Mixedwood to Foothills lakes. Similar relationships have been reported for MeHg concentrations in zooplankton (Larsson et al. 1992; Westcott and Kalff 1996) and larger fish species (Rodgers and Beamish 1983; Grieb et al. 1990; Driscoll et al. 1994). MeHg concentrations in *C. shurtleffi*, *Aeshna* spp., and *C. inconstans* increased by 1.6- to 1.8-fold per unit decrease in pH, compared to a 1.4-fold increase reported for sunfish (*Lepomis gibbosus*) in southern Ontario lakes (Wren and MacCrimmon 1983). Low pH may promote Hg bioavailability (Barkay et al. 1997) and methylation rates (Xun et al. 1987; Miskimmin et al. 1992) by reducing the binding capacity of organic ligands such as DOC (Wetzel 2001). Lake water pH, hardness, and DOC concentrations were all particularly high in the productive Mixedwood lakes and thus their inhibitory effects on Hg methylation and MeHg availability were likely compounded. MeHg concentrations in biota were not correlated with lake water colour or DOC concentration because the proportion of organic carbon from autochthonous and allochthonous sources was not consistent among lakes.

3.4.3 Biomagnification of MeHg in the littoral food web

The influence of productivity on MeHg bioaccumulation appears to involve an interaction between the effects of lake chemistry on MeHg uptake by primary producers, and biodilution from high growth rates and biomass throughout the food web. Because high productivity is generally accompanied by lake water conditions that are inhibitory to MeHg accumulation, such as high pH, it is difficult to determine the relative significance

of biodilution to contaminant concentrations (e.g., Larsson et al. 1992; Kidd et al. 1999). Results from enclosure experiments by Rudd and Turner (1983) suggest that elevated pH of lake water, not biodilution, is the primary factor affecting Hg concentrations in productive systems. In my lakes, however, biomagnification of MeHg appeared to be less efficient in mesotrophic lakes than in oligo-mesotrophic lakes. Because MeHg uptake occurs primarily through the food chain, the direct effects of lake chemistry on Hg and MeHg bioavailability cannot explain the apparent change in transfer efficiency of MeHg among secondary consumers. Thus high growth rates and biomass in productive lakes may dilute MeHg concentrations at each step in the food chain and cause an apparent decrease in biomagnification over multiple trophic levels.

The average slope of the MeHg vs. $\delta^{15}N$ relationships (0.2) of my lakes was at the low margin of the range of biomagnification rates determined for pelagic food webs from less productive systems (0.2 to 0.3; e.g. Kidd et al. 1995; Atwell et al. 1998; Bowles et al. 2001). On average, MeHg concentrations increased 5-fold per trophic level in the Swan Hills food webs, compared to ~10- to 15-fold per trophic level reported for pelagic food webs on the Boreal Shield (Plourde et al. 1997). This difference may be only partially explained by the difference in lake trophic status between these two groups of lakes. Biomagnification rates determined for pelagic food webs may be exaggerated by the bioaccumulative MeHg burdens of longer-lived fish species. All the biota collected from my lakes had relatively similar 1- to 2-yr lifecycles, thus biomagnification was less affected by time of exposure to MeHg and bioaccumulative MeHg burdens.

Variation in MeHg bioaccumulation among systems may also reflect the influence of water chemistry and trophic status on food web structure. In Swedish lakes, water column pH shifted the prey selection of northern pike (*Esox lucius*) from acid-sensitive roach (*Rutilus rutilus*) to perch (*Perca fluviatalis*), a trophically-elevated prey species with higher MeHg concentrations than roach (Bjorklund et al. 1984). Similarly, the acid-sensitive detritivore *G. lacustris* was abundant in productive, alkaline lakes in Mixedwood watersheds, but absent from the relatively acidic lakes in the Foothills. This may have driven predators such as *C. shurtleffi*, *Aeshna* spp., and *C. inconstans* in Foothills lakes to select prey from higher in the food web (e.g., Pritchard 1964), and thereby increase their exposure to MeHg. Lake-to-lake comparisons of MeHg

concentrations in macroinvertebrates may also be complicated by flexibility in feeding habits within taxa. For example, MeHg concentrations in Trichoptera (*Agrypnia* spp. and *Phryganea* spp.) from my lakes were 4-fold lower than in *Agrypnia* spp. collected from a Boreal Shield lake, a difference too large to be explained by lake chemistry (Tremblay et al. 1996). The difference in MeHg concentrations in *Agrypnia* spp. between Boreal Plain and Boreal Shield lakes may thus reflect different prey selection. Based on $\delta^{13}C$ signatures, Trichoptera in my lakes apparently fed primarily on aquatic macrophytes, typically a low MeHg food source. In less alkaline lakes, Trichoptera $\delta^{13}C$ appeared to shift towards the signatures of other consumers, and their baseline adjusted $\delta^{15}N$ increased, indicative of increased trophic position. *Agrypnia* spp. are also known to feed on aquatic insects and crustaceans (Wiggins 1998) and may shift their diet according to the lake trophic status and resource availability.

Taxon-specific factors such as feeding strategy appeared to influence the relationship between MeHg concentrations and trophic position. In particular, *Notonecta* spp. appeared to accumulate MeHg more efficiently than other taxa (Fig. 3-8). *Notonecta* spp. are morphologically adapted to pierce the exoskeletons of their prey and feed on liquefied inner tissues (Merrit and Cummins 1996). The feeding strategy of *Notonecta* spp. appears to provide a greater food conversion efficiency than for shredders, collectors, or engulfers (Cummins and Klug 1979) and likely increases assimilation of MeHg in *Notonecta* spp. relative to organisms at higher trophic positions. High C:N ratios in *Notonecta* spp. indicate a higher lipid content than the other taxa (Anderson 1994) and may reflect differences in feeding strategies.

3.4.4 Swan Hills Treatment Centre (SHTC) lakes

There are no published studies that link high MeHg concentrations in fish from lakes located near the SHTC to emissions from the facility. MeHg concentrations in taxa from Edith Lake, located 10 km upwind from the SHTC, were high relative to the other study lakes, but were similar to, or below, values reported from lakes with comparable water chemistry on the Boreal Shield (Tremblay et al. 1996; Kidd et al. 1999). The water of Edith Lake was characterized by relatively moderate productivity, low pH, and high colour. A large bog (17% of the drainage basin) located on the south-east shore of the lake was the primary source of colour and acidity (e.g., Halsey et al. 1997), and a likely

source of MeHg (e.g., St. Louis et al. 1994). The pH of Chrystina Lake, located 1.5 km from the SHTC, was acidic prior to operation of the plant (1985 pH = 6.8; Stanley Associates Engineering Ltd. 1990) and has not decreased following 13 yr of plant operations (2000 pH = 7.1). Aquatic biota in Edith and Chrystina lakes were trophically elevated relative to biota in other burned and reference lakes, and by feeding at a higher trophic position may accumulate more MeHg (Cabana et al. 1994). Near-site deposition of Hg from the SHTC did not differ from background deposition during the first 3 yr of plant operation, when high Hg concentrations were detected in Chrystina Lake brook trout (Stanley Associates Engineering Ltd. 1990). Even if emissions of Hg from the SHTC have increased in subsequent years, near-site deposition of Hg may not translate into higher concentrations in fish (Pinkney et al. 1997). Thus, high MeHg concentrations in the SHTC lakes appear to be explained in large part by natural lake chemistry, food web structure, and watershed features.

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Table 3-1. Distribution of macroinvertebrates and fish in the study lakes.

Taxon	Number of lakes in which taxa occur			
	Reference		Burned	SHTC
	Mixedwood	Foothills	Foothills	Foothills
Trichoptera	3	2	4	1
Gammarus lacustris	3	2	3	-
Cordulia shurtleffi	2	1	3	2
Aeshna spp.	2	1	4	2
Notonecta spp.	3	2	5	2
Culaea inconstans	3	1	2	2
Margariscus margarita	1	-	-	-

Note: SHTC, Swan Hills Treatment Centre.

Table 3-2. Watershed and lake characteristics and July to August mean values for water quality.

Lake	Ecoregion	LA (ha)	CA/LV (m^{-1})	Z_m (m)	ELEV (m)	CON (%)	WET (%)	BURN (%)	TP ($\mu g \cdot L^{-1}$)	DOC ($mg \cdot L^{-1}$)	Colour ($mg\ Pt \cdot L^{-1}$)	pH	COND ($\mu S \cdot cm^{-1}$)	ALK ($mg \cdot L^{-1}$)
Burned														
Mons	Foothills	19	2.6	1.1	990	90	7	90	28	12	36	8.7	82	47
Twin	Foothills	13	7.8	1.4	1040	88	10	70	23	17	139	6.8	28	12
Shelly	Foothills	23	2.8	2.0	1100	57	11	80	24	9	29	7.5	88	33
Michel	Foothills	13	7.2	1.4	990	63	10	50	19	16	71	7.4	61	33
Swartz	Foothills	28	5.4	2.2	1090	86	7	20	31	12	40	8.0	94	69
Mean		19	5.2	1.6	1042	77	9	62	25	**13**	63	7.3	71	39
Reference														
Rat	Foothills	10	13.0	1.0	1095	80	9		24	11	34	8.1	84	47
Marigold	Foothills	34	1.8	1.5	1115	87	13		16	6	12	7.7	57	32
Christina	Foothills	39	1.7	3.9	1015	95	4		20	8	17	7.1	32	17
Edith	Foothills	22	2.8	1.3	995	80	17		16	11	46	6.6	64	8.7
Ref. Foothills Mean		26	4.8	1.9	1055	88	12		19	9	27	7.0	59	26
Pleasant	Mixwood	18	6.9	2.0	775	4	2		34	14	16	9.4	138	87
Goodwin	Mixwood	18	6.5	2.2	860	32	3		32	12	41	8.1	202	118
Delorme	Mixwood	34	4.8	2.1	610	40	10		38	32	72	8.6	246	122
Ref. Mixedwood Mean		23	6.1	2.1	**748**	**25**	5		**35**	**19**	43	**8.5**	**195**	**109**
Overall mean		22	5.2	1.8	973	67	9	-	25	13	46	7.8	98	52

Note: LA, lake area; Z_m, mean depth; ELEV, elevation; CON, % conifer in drainage basin; WET, % wetland cover; BURN, % burned; TP, total phosphorus; DOC, dissolved organic carbon; COND, conductivity; ALK, alkalinity; means in bold are different relative to reference-Foothills means (Mann-Whitney, $P \leq 0.05$).

Table 3-3. Pre- and post-disturbance $\delta^{13}C$, $\delta^{15}N$, and body length of Delorme Lake macroinvertebrates and fish.

Taxon	*n*		$\delta^{13}C$ (‰)		$\delta^{15}N$ (‰)		Body length (mm)	
	2000	2001	2000	2001	2000	2001	2000	2001
Agrypnia spp.	12	12	-16.1±0.2	-16.5±0.2	-0.6±0.3	**0.5± 0.2**	17.2±0.2	**18.3±0.4**
Gammarus lacustris	12	12	-23.9±0.1	-23.9±0.1	3.3±0.1	**3.8±0.1**	6.2±0.2	**7.0±0.1**
Cordulia shurtleffi	12	10	-27.2±0.2	**-25.9±0.4**	4.3±0.1	**4.0±0.3**	20.5±0.2	20.0±0.5
Notonecta spp.	12	9	-24.7±0.6	-24.9±0.9	4.9±0.2	4.6±0.2	11.2±0.1	11.6±0.2
Margariscus margarita	12	12	-26.5±0.2	-26.5±0.1	8.2±0.1	8.2±0.1	63.6±3	63.5±0.7
Culaea inconstans	12	11	-26.0±0.2	-25.1±0.2	9.2±0.1	9.1±0.1	54.6±0.8	55.0±1.2

Note: Bolded means are different from pre-disturbance means (*t*-test; $P \leq 0.05$).

Table 3-4. Stable isotope ratios and body lengths in macroinvertebrates and fish.

Lake	*n*	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)	TPOS	LGTH (mm)
Burned lakes					
Trichoptera	4	-21.9±2.4	0.3±0.8	1.4±0.2	19.0±2.2
G. lacustris.	3	-23.7±1.6	2.5±0.8	2.0±0.2	5.6±0.2
C. shurtleffi	3	-26.5±1.5	2.4±0.2	2.1±0.0	20.3±0.1
Aeshna spp.	4	-26.1±1.1	3.8±0.4	2.4±0.1	34.3±0.9
Notonecta spp.	5	-26.0±0.9	3.9±0.1	2.4±0.0	11.9±0.2
C. inconstans	2	-23.8±1.2	7.0±0.2	3.4±0.0	53.2±8.1
Reference lakes					
Trichoptera	5	-18.4±1.2	-0.4±0.6	1.2±0.2	17.8±0.8
G. lacustris	4	-23.5±1.7	1.9±0.5	1.9±0.1	6.1±0.2
C. shurtleffi	3	-26.5±0.9	3.2±0.6	2.1±0.0	20.3±0.3
Aeshna spp.	3	-22.1±0.1	9.2±0.3	3.6±0.0	32.2±1.2
Notonecta spp.	5	-25.3±1.0	3.9±0.4	2.5±0.0	11.5±0.1
C. inconstans	4	-22.9±2.0	8.2±0.4	3.6±0.1	54.9±1.4
SHTC lakes					
Trichoptera	1	-25.6	5.4	2.3	16.3
G. lacustris					
C. shurtleffi	2	-28.1±0.8	4.7±0.7	2.3±0.0	21.1±0.1
Aeshna spp.	2	-26.4±1.5	6.0±0.4	2.7±0.1	37.4±1.7
Notonecta spp.	2	-26.4±0.2	4.4±0.0	2.3±0.2	11.6±1.7
C. inconstans	2	-24.0±1.1	8.3±0.5	3.6±0.1	54.8±1.7

Note: mean ± standard error; TPOS, trophic position; LGTH, body length; SHTC, Swan Hills Treatment Centre.

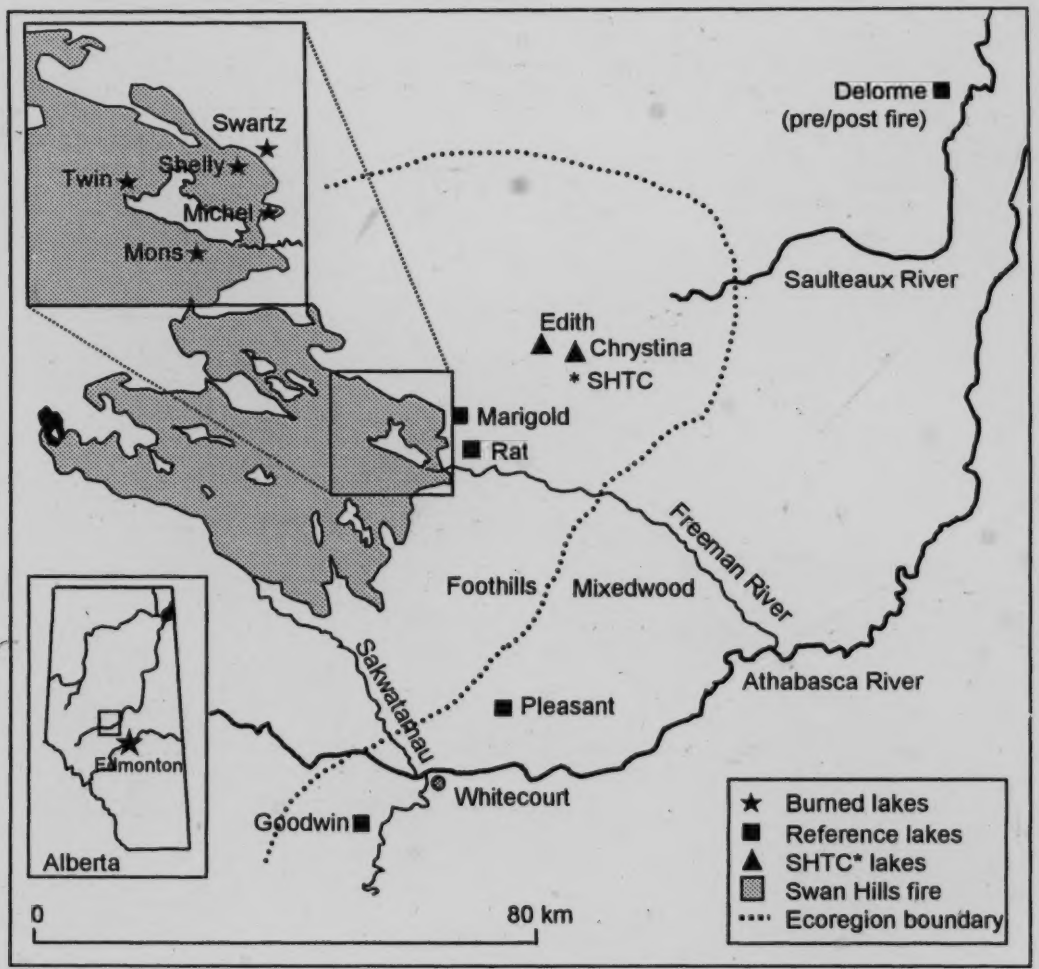

Figure. 3-1. Swan Hills study lakes (*SHTC, Swan Hills Treatment Centre).

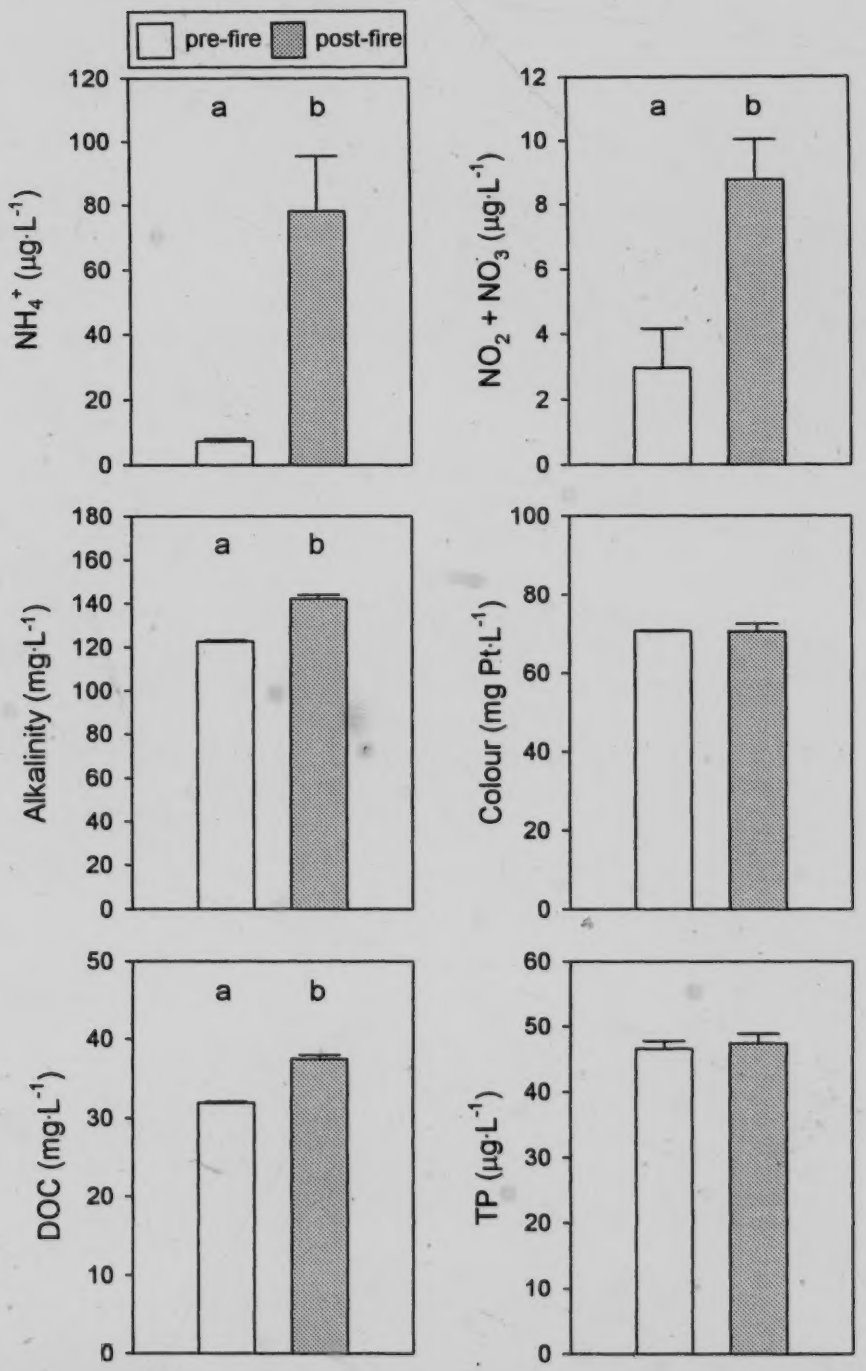

Figure 3-2. Pre-fire vs. post-fire comparison of August to October means for water quality in Delorme Lake (whiskers represent 1 standard error of the mean; letters indicate significant differences between pre- and post-fire years; Mann-Whitney *U*-test, $P \leq 0.05$; pre-fire $n = 4$, post-fire $n = 8$).

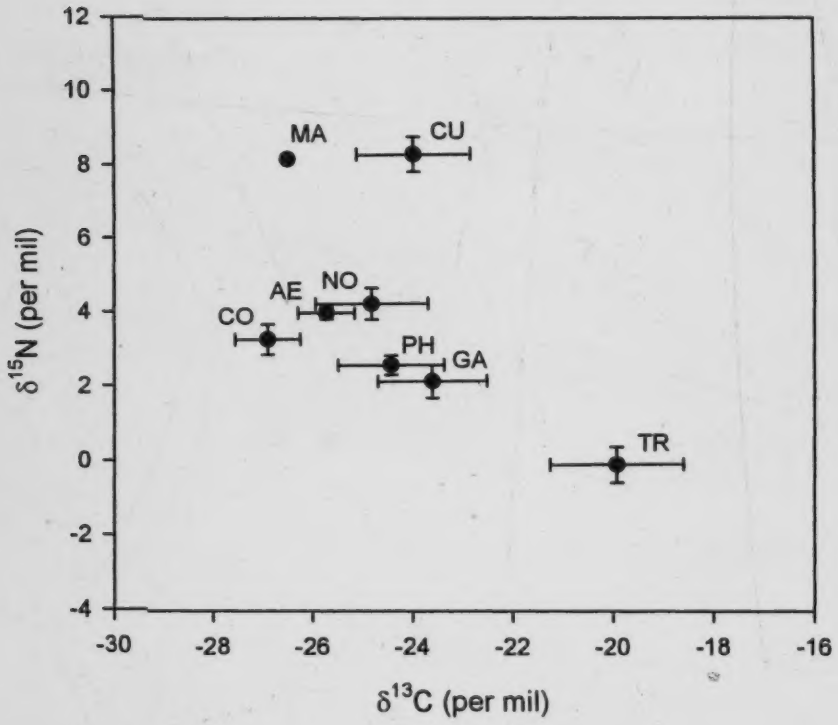

Figure 3-3. Overall mean $\delta^{15}N$ and $\delta^{13}C$ signatures in macroinvertebrates and fish (whiskers represent ±1 standard error; TR, Trichoptera, GA, *Gammarus lacustris*; PH, *Physa* spp.; CO, *Cordulia shurtleffi*; AE, *Aeshna* spp., NO, *Notonecta* spp.; CU, *Culaea inconstans*; MA, *Margariscus margarita*).

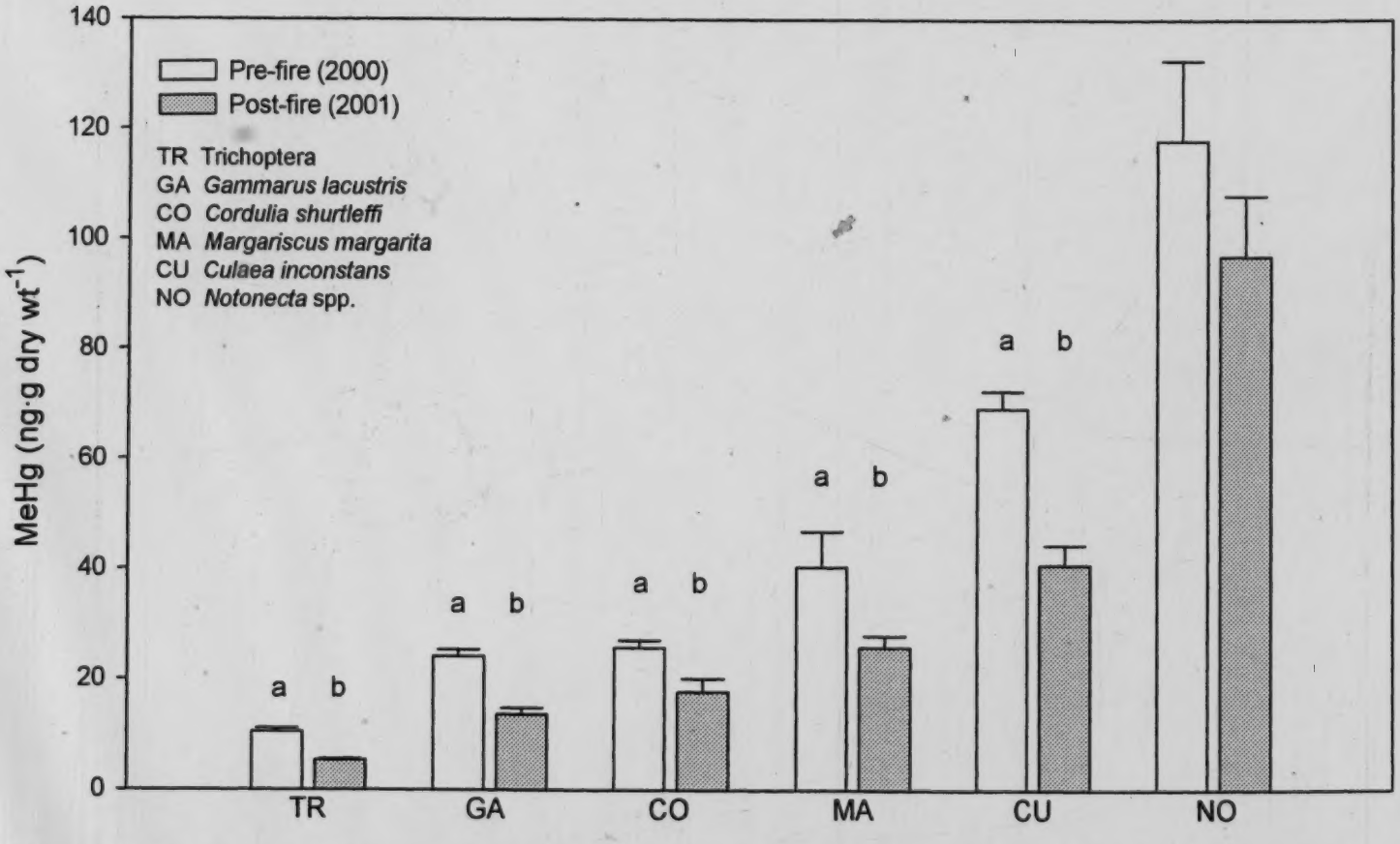

Figure 3-4. Pre-fire vs. post-fire comparison of MeHg concentrations in macroinvertebrates and fish from Delorme Lake (whiskers represent 1 standard error of the mean; letters indicate significant difference between years; *t*-test, $P < 0.05$).

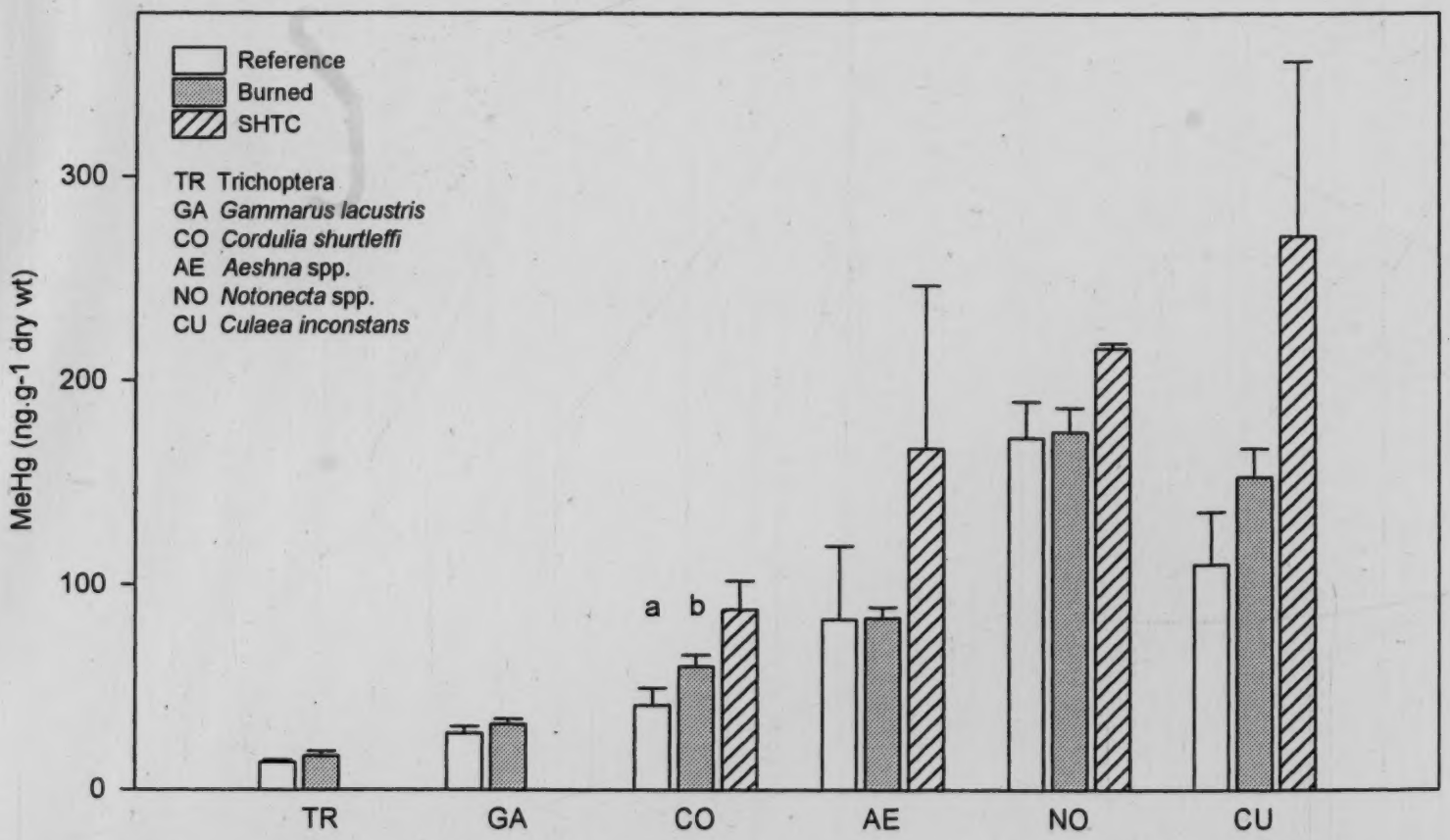

Figure 3-5. MeHg concentrations in macroinvertebrates and fish from reference, burned and SHTC (Swan Hills Treatment Centre) lakes (error bars = 1 standard error, letters indicate significant difference between burned and reference, $P < 0.05$, Mann-Whitney U test).

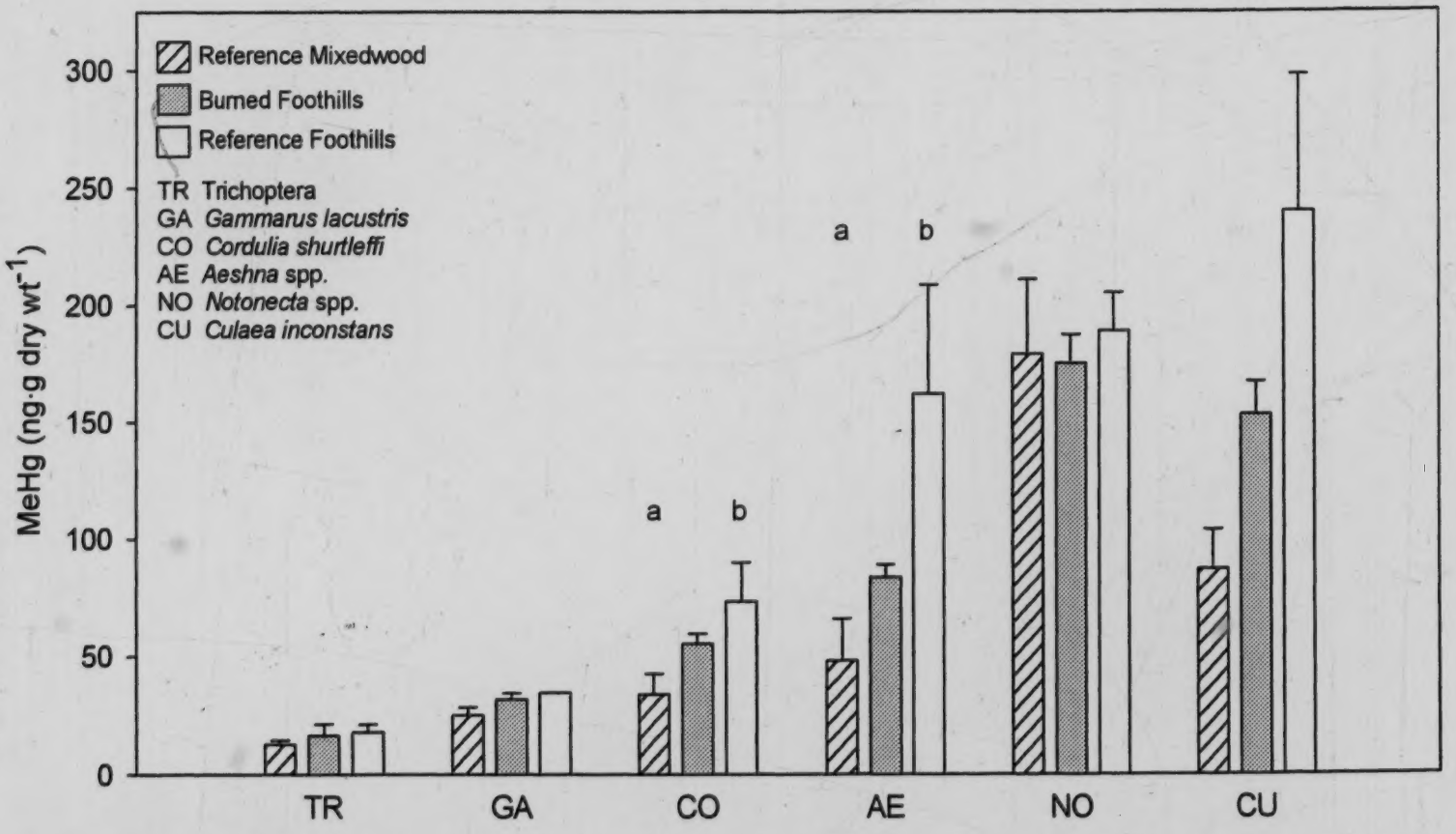

Figure 3-6. Comparison of MeHg concentrations in macroinvertebrates and fish from burned Foothills, reference Foothills, and reference Mixedwood lakes (error bars = 1 standard error; letters indicate significant differences; Mann-Whitney *U*-test, $P \leq 0.05$).

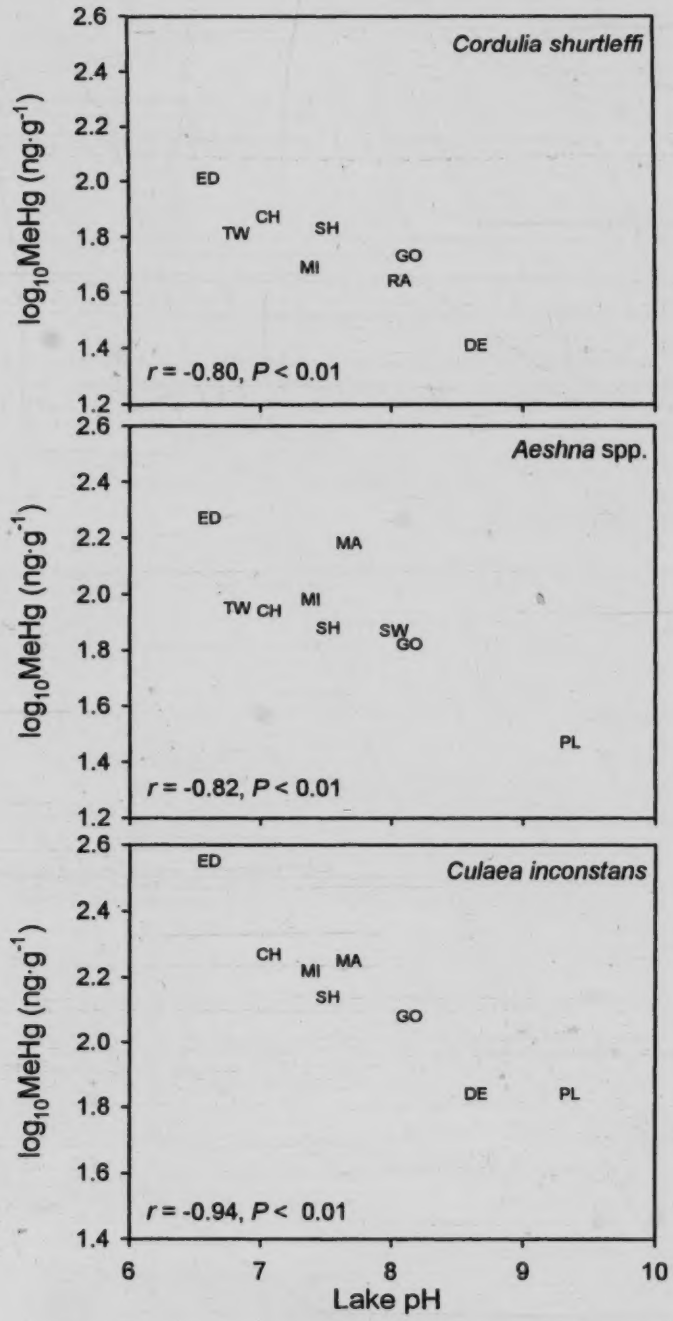

Figure 3-7. Scatterplots of MeHg concentration and lake pH in dragonfly larvae (*Cordulia shurtleffi* and *Aeshna* spp.) and brook stickleback (*Culaea inconstans*) (lake codes: MO, Mons; TW, Twin; SH, Shelly; MI, Michel; SW, Swartz; RA,Rat; MA, Marigold; PL, Pleasant; GO, Goodwin; DE, Delorme; ED, Edith; CH, Chrystina).

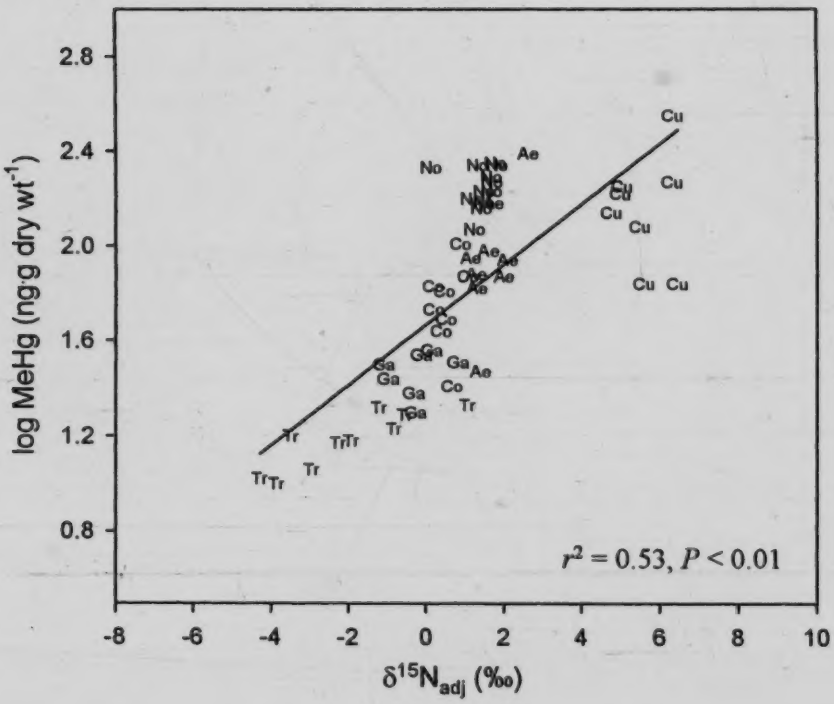

Figure 3-8. Relationship between MeHg concentration and baseline-adjusted $\delta^{15}N$ (Tr, Trichoptera; Ga, *Gammarus lacustris*; Co, *Cordulia shurtleffi*; Ae, *Aeshna* spp.; No, *Notonecta* spp.; Cu, *Culaea inconstans*).

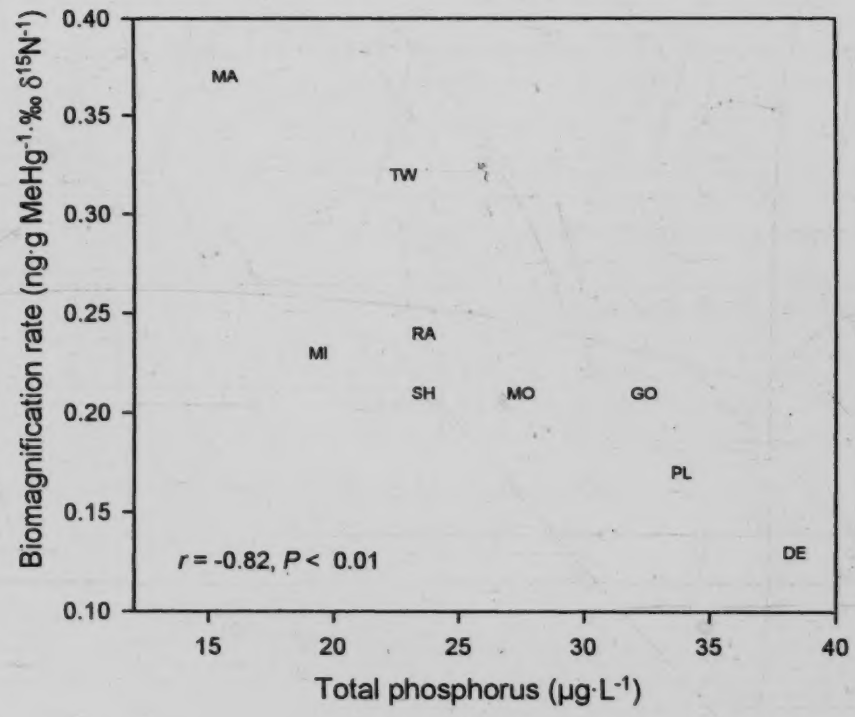

Figure 3-9. Scatterplot of biomagnification rate (the slope from the $\log_{10}$MeHg vs. $\delta^{15}N$ relationship in each lake) and total phosphorus concentration (Lake codes: DE, Delorme; PL, Pleasant; GO, Goodwin; RA, Rat; MO, Mons; SH, Shelly; MI, Michel; TW, Twin; MA, Marigold).

Chapter 4. GENERAL DISCUSSION AND CONCLUSIONS

MeHg concentrations in aquatic biota did not increase after forest fire in the Boreal Plain watersheds of the Swan Hills region. My results are consistent with the view that the effect of forest disturbance on MeHg bioaccumulation in lakes depends largely on the post-disturbance export of terrestrial organic matter and its impacts on water chemistry. Post-disturbance increases in MeHg concentrations in biota are more likely to occur when watershed inputs of allochthonous DOC are sufficient to increase lake water colour and acidity, and suppress primary production. Post-fire inputs of allochthonous DOC did not occur in my study area, thus MeHg concentrations in aquatic biota remained constant or decreased.

Post-disturbance export of allochthonous organic matter will vary according to disturbance type (i.e., logging vs. fire), catchment soils, and vegetation. Unlike logging, forest fire may volatilize soil organic matter and reduce the export of allochthonous DOC (Carignan et al. 2000). The Delorme Lake fire was particularly intense, and ground cover in the catchment was reduced to exposed mineral soil after the fire. Similarly, forest fire all but removed the thin soil cover overlying bedrock in the Boreal Shield catchments studied by Garcia and Carigan (2000), where there was no post-fire increase in lake water DOC concentrations or MeHg concentrations in aquatic biota. Alternatively, where soil organic layers are well-developed (e.g., peatlands), post-fire lake inputs of allochthonous DOC may be sufficient to elevate colour and acidity in lake water (McEachern et al. 2000) and increase MeHg concentrations in biota.

In catchments dominated by deciduous vegetation, such as Delorme Lake, post-disturbance export of humic substances may be low regardless of disturbance type, due to low DOC concentrations in the soils (Cronan and Aiken 1987). Logging <40% of deciduous-dominated Mixedwood watersheds resulted in no increase in allochthonous DOC inputs or colour in lake water (Prepas et al. 2001). Logging in conifer-dominated watersheds on the Boreal Shield increased DOC concentrations and color in lake water and MeHg concentrations in aquatic biota (Garcia and Carignan 2000). The effects of clear-cutting on Boreal Plain lakes remain poorly understood, however these findings suggest that conifer-dominated Boreal Foothills lakes may be more susceptible to a post-

logging increase in MeHg concentrations in biota, than lakes in deciduous-dominated watersheds.

Detection of a lake water response to forest disturbance depends on the timeframe of the study. Three months post-fire, dissolved inorganic nitrogen (DIN) concentrations had increased 10-fold in Delorme Lake water, whereas two years post-fire in the Boreal Foothills there was no difference in DIN concentrations between reference and burned lakes. The burned Foothills lakes were sampled in the third summer following a spring fire, by which time labile inorganic N had likely been taken up by recolonizing vegetation. Also, annual precipitation was below average in the Boreal Foothills during the first two summers post-fire, and may have further reduced export of soil nutrients. Year-to-year variation in precipitation may be an important determinant of lake water chemistry and algal biomass in sedimentary basins such as the Boreal Plain (Charette 2001), and could obscure post-disturbance changes in lake chemistry and MeHg concentrations.

My study reinforced three key aspects of MeHg bioaccumulation in benthic macroinvertebrates. First, MeHg concentrations in benthic macroinvertebrates reflect open-water conditions (i.e., pH and productivity) that also influence MeHg concentrations in pelagic plankton and fish species (McMurtry et al. 1989; Westcott and Kalff 1996). The relationships between water chemistry and MeHg concentrations in my lakes were particularly strong for predators such as *C. shurtleffi* and *Aeshna* spp. Second, MeHg concentrations in macroinvertebrates were related to trophic position, as defined by $\delta^{15}N$, although the relationship may be influenced by taxon-specific feeding strategies. Further, lake-to-lake variation in MeHg concentrations within taxa may reflect omnivory and flexible feeding habits in macroinvertebrates.

MeHg concentrations in macroinvertebrates and baitfish of the Swan Hills region were generally below the range of values reported for these organisms on the Boreal Shield (Tremblay et al. 1996; Hall et al. 1998; Kidd et al. 1999). Within the Swan Hills, lake productivity and pH varied along an elevational gradient between Boreal Mixedwood and Boreal Foothills ecoregions, and strongly influenced MeHg concentrations. My study emphasizes the importance of ecoregional factors when studying the effects of watershed disturbance on MeHg bioaccumulation. Overall, my

data provide a framework for elucidating the relationships between watershed characteristics, lake chemistry, trophic ecology, and MeHg bioaccumulation in lakes in sedimentary basins.

4.1 Literature cited

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Appendix A:

Summary data for MeHg concentration, stable isotope ratios, trophic position, and body length.

Table A1. Summary data for MeHg concentration, stable isotope ratios, trophic position, and body length: Trichoptera.

Lake	Genus	*n*	MeHg (ng·g dry wt^{-1})	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)	TPOS	C:N	LGTH (mm)
Burned lakes								
Mons	*Phryganea* spp.	10	20.8±1.3	-17.4±0.1	1.4±0.2	1.6±0.1	4.8±0.1	19.0±0.6
Twin	*Phryganea* spp.	6	19.4±2.6	-27.0±0.2	1.5±0.2	1.8±0.0	4.6±0.1	13.8±0.7
Shelly	*Phryganea* spp.	10	9.9±1.1	-18.3±0.2	-1.9±0.1	0.9±0.0	4.8±0.1	24.6±0.8
Michel	*Agrypnia* spp.	3	15.1±3.3	-24.9±0.4	0.3±0.3	1.4±0.1	5.0±0.1	18.6±1.7
Mean		4	16.3±2.4	-21.9±2.4	0.3±0.8	1.4±0.2	4.8±0.1	19.5±1.8
Reference lakes								
Rat	*Phryganea* spp.	10	14.9±1.1	-19.8±0.2	-0.3±0.2	1.3±0.1	4.7±0.1	18.9±0.5
Marigold	*Agrypnia* spp.	10	17.0±3.4	-19.2±0.4	1.8±0.2	1.8±0.1	4.8±0.1	14.9±0.3
Pleasant	*Phryganea* spp.	10	11.6±1.0	-15.0±0.1	-2.0±0.2	1.1±0.1	4.5±0.0	19.6±0.5
Goodwin	*Agrypnia* spp.	12	16.0±4.3	-21.7±0.6	-0.7±0.3	1.0±0.1	4.5±0.0	18.4±1.7
Delorme	*Phryganea* spp.	12	10.4±0.6	-16.1±0.2	-0.6±0.3	0.7±0.1	5.1±0.0	17.2±0.2
Mean		5	14.0±1.3	-18.4±1.2	-0.4±0.6	1.2±0.2	4.7±0.1	17.8±0.8
SHTC lakes								
Christina	*Agrypnia* spp.	2	21.4±2.4	-25.6±1.0	5.4±0.1	2.3±0.0	4.4±0.1	16.3±0.3
Overall mean		10	15.6±1.3	-20.5±1.3	0.5±0.7	1.4±0.2	4.7±0.1	18.1±0.8

Note: TPOS, trophic position; LGTH, length; SHTC, Swan Hills Treatment Centre.

Table A2. Summary data for MeHg concentration, stable isotope ratios, trophic position, and body length: *Gammarus lacustris*.

Lake	*n*	MeHg (ng·g dry wt^{-1})	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)	TPOS	C:N	LGTH (mm)
Burned lakes							
Mons	10	36.0±1.9	-20.6±0.1	2.8±0.1	2.0±0.0	5.2±0.1	5.4±0.0
Shelly	10	27.3±1.2	-24.6±0.2	1.0±0.1	1.7±0.0	4.6±0.0	6.0±0.1
Swartz	10	32.2±1.5	-25.9±0.1	3.8±0.1	2.2±0.0	4.5±0.0	5.5±0.0
Mean (burned)	3	31.8±2.5	-23.7±1.6	2.5±0.8	2.0±0.2	4.8±0.2	5.6±0.2
Reference lakes							
Rat	10	34.5±1.2	-23.5±0.3	1.8±0.1	2.0±0.0	4.7±0.0	5.6±0.0
Pleasant	10	19.8±1.0	-19.2±0.1	0.7±0.1	1.9±0.0	4.6±0.0	6.4±0.1
Goodwin	11	31.4±1.5	-27.5±0.1	1.7±0.1	1.7±0.0	4.7±0.1	6.2±0.0
Delorme	12	24.1±1.2	-23.9±0.1	3.3±0.1	1.9±0.0	4.7±0.0	6.2±0.2
Mean (reference)	4	27.4±3.4	-23.5±1.7	1.9±0.5	1.9±0.1	4.7±0.02	6.1±0.2
Overall mean	7	29.3±2.2	-23.6±1.1	2.2±0.4	1.9±0.1	4.7±0.1	5.9±0.1

Note: TPOS, trophic position; LGTH, length.

Table A3. Summary data for MeHg concentration, stable isotope ratios, trophic position, and body length: *Cordulia shurtleffi*.

Lake	*n*	MeHg (ng·g dry wt^{-1})	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)	TPOS	C:N	LGTH (mm)
Burned lakes							
Twin	10	63.9±4.4	-29.5±0.4	2.5±0.1	2.1±0.0	4.4±0.0	20.2±0.3
Shelly	8	52.8±3.3	-24.6±0.4	2.1±0.1	2.1±0.0	4.7±0.1	20.4±0.5
Michel	10	49.1±2.4	-25.5±0.4	2.7±0.1	2.2±0.0	4.4±0.0	20.3±0.2
Mean (burned)	3	55.3±4.4	-26.5±1.5	2.4±0.2	2.1±0.0	4.5±0.1	20.3±0.1
Reference lakes							
Rat	10	43.9±3.6	-24.6±0.2	2.3±0.1	2.1±0.0	4.6±0.1	19.7±0.2
Goodwin	11	43.0±2.8	-27.6±0.2	3.0±0.1	2.1±0.0	4.8±0.1	20.6±0.2
Delorme	12	25.7±1.4	-27.2±0.2	4.3±0.1	2.2±0.0	4.6±0.1	20.5±0.2
Mean (reference)	3	37.5±5.9	-26.4±0.9	3.2±0.6	2.1±0.0	4.7±0.1	20.3±0.3
SHTC lakes							
Edith	10	102.1±9.5	-28.9±0.8	4.0±0.2	2.3±0.0	4.4±0.3	21.2±0.3
Christina	3	73.6±6.3	-27.3±0.1	5.4±0.4	2.3±0.1	4.4±0.1	21.0±0.0
SHTC mean	2	87.9±14.3	-28.1±0.8	4.7±0.7	2.3±0.0	4.4±0.0	21.1±0.1
Overall mean	8	56.8±8.2	-26.9±0.6	3.3±0.4	2.2±0.0	4.5±0.1	20.5±0.2

Note: TPOS, trophic position; LGTH, length; SHTC, Swan Hills Treatment Centre.

Table A4. Summary data for MeHg concentration, stable isotope ratios, trophic position, and body length: *Aeshna* spp.

Lake	*n*	MeHg (ng·g dry wt^{-1})	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)	TPOS	C:N	LGTH (mm)
Burned lakes							
Twin	6	90.0±10.9	-28.2±0.8	3.2±0.2	2.3±0.0	4.0±0.1	31.7±1.0
Shelly	5	76.1±11.6	-23.2±1.0	3.2±0.2	2.4±0.1	4.3±0.2	34.8±0.9
Michel	3	95.8±24.3	-25.7±0.8	3.8±0.5	2.5±0.2	4.4±0.3	35.3±2.9
Swartz	3	73.4±9.2	-27.3±0.5	5.0±0.1	2.6±0.0	4.3±0.1	35.3±2.6
Mean (burned)	4	83.8±5.4	-26.1±1.1	3.8±0.4	2.4±0.1	4.3±0.1	34.3±0.9
Reference lakes							
Marigold	2	105.0±48.0	-21.2±0.6	4.4±0.2	2.1±0.0	4.5±0.3	32.4±8.2
Pleasant	4	29.7±1.7	-18.2±0.3	2.4±0.6	2.1±0.0	4.1±0.1	30.0±3.7
Goodwin	3	66.6±7.0	-26.9±0.6	4.1±0.1	2.2±0.0	4.5±0.2	34.3±1.3
Mean (reference)	3	83.1±36.5	-22.1±0.1	3.6±0.6	2.1±0.0	4.4±0.1	32.2±1.2
SHTC lakes							
Edith	5	247.0±63.5	-27.9±1.0	5.7±0.3	2.8±0.2	4.5±0.1	35.8±0.6
Christina	8	86.3±7.0	-24.9±0.2	6.4±0.2	2.6±0.1	4.4±0.1	39.1±1.6
SHTC mean	2	166.7±80.4	-26.4±1.5	6.0±0.4	2.7±0.1	4.4±0.0	37.4±1.7
Overall mean	9	102.0±21.1	-24.8±1.1	4.2±0.4	2.4±0.0	4.3±0.1	34.3±0.9

Note: TPOS, trophic position; LGTH, length; SHTC, Swan Hills Treatment Centre.

Table A5. Summary data for MeHg concentration, stable isotope ratios, trophic position, and body length: *Notonecta* spp.

Lake	*n*	MeHg (ng·g dry wt^{-1})	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)	TPOS	C:N	LGTH (mm)
Burned lakes							
Mons	4	143.2±14.1	-22.6±0.8	4.1±0.4	2.4±0.1	4.6±0.2	12.2±0.3
Twin	10	171.2±11.9	-27.6±0.6	3.7±0.1	2.5±0.0	5.3±0.2	11.9±0.1
Shelly	10	216.6±21.2	-27.8±0.6	3.8±0.2	2.5±0.1	5.7±0.1	11.2±0.1
Michel	10	184.2±27.6	-25.8±0.9	3.9±0.3	2.5±0.1	5.4±0.2	12.0±0.1
Swartz	10	159.2±23.7	-26.1±0.9	4.2±0.2	2.3±0.0	6.0±0.1	12.0±0.1
Mean (burned)	5	174.9±12.4	-26.0±0.9	3.9±0.1	2.4±0.0	5.4±0.2	11.9±0.2
Reference lakes							
Rat	10	152.4±18.5	-24.8±0.6	3.4±0.2	2.4±0.1	5.1±0.3	11.9±0.2
Marigold	10	171.3±19.3	-26.7±1.0	4.2±0.3	2.4±0.1	6.2±0.1	11.2±0.1
Pleasant	8	193.7±39.1	-21.7±1.5	2.7±0.5	2.9±0.1	5.3±0.2	11.4±0.1
Goodwin	10	226.0±31.7	-28.0±0.4	4.5±0.2	2.5±0.0	5.4±0.1	11.8±0.2
Delorme	11	118.0±14.4	-24.7±0.6	4.9±0.2	2.4±0.0	5.6±0.2	11.2±0.0
Mean (reference)	5	172.3±18.3	-25.2±1.0	3.9±0.4	2.5±0.0	5.5±0.2	11.5±0.1
SHTC lakes							
Edith	10	218.8±12.8	-26.6±0.5	4.4±0.1	2.4±0.0	5.8±0.1	11.6±0.1
Christina	10	214.0±17.9	-26.3±0.5	4.5±0.2	2.2±0.0	5.9±0.1	11.6±0.1
SHTC mean	2	216.4±2.4	-26.4±0.2	4.4±0.0	2.3±0.2	5.9±0.0	11.6±1.7
Overall mean	12	180.7±9.9	-25.7±0.6	4.0±0.2	2.4±0.0	5.5±0.1	11.7±0.1

Note: TPOS, trophic position; LGTH, length; SHTC, Swan Hills Treatment Centre.

Table A6. Summary data for MeHg concentration, stable isotope ratios, trophic position, and body length: *Culaea inconstans*.

Lake	*n*	MeHg (ng·g dry wt^{-1})	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)	TPOS	C:N	LGTH (mm)
Burned lakes							
Shelly	8	139±6.5	-22.6±1.0	6.7±0.1	3.4±0.0	3.4±0.0	61.4±2.9
Michel	6	167.3±25.6	-25.0±1.0	7.2±0.1	3.5±0.2	3.4±0.0	45.0±0.6
Mean (burned)	2	153.2±14.2	-23.8±1.2	7.0±0.2	3.4±0.0	3.4±0.0	53.2±8.1
Reference lakes							
Marigold	8	177.8±19.9	-18.2±0.2	7.7±0.3	3.5±0.1	3.5±0.0	58.4±3.4
Pleasant	10	69.7±5.8	-20.8±0.1	7.4±0.1	3.9±0.0	3.4±0.0	51.7±1.3
Goodwin	10	121.5±3.9	-26.5±0.2	8.3±0.2	3.6±0.0	3.4±0.0	55.0±0.9
Delorme	12	69.3±3.1	-26.0±0.2	9.2±0.3	3.6±0.0	3.4±0.0	54.6±0.8
Mean (reference)	4	109.6±25.8	-22.9±2.0	8.2±0.4	3.7±0.1	3.4±0.0	54.9±1.4
SHTC lakes							
Edith	10	356.7±13.4	-27.9±0.2	9.4±0.2	3.9±0.0	3.4±0.0	58.0±1.9
Christina	10	185.7±14.2	-24.9±0.2	10.7±0.2	5.1±0.1	3.4±0.0	54.4±1.5
SHTC mean	2	271.2±85.5	-26.4±1.5	10.0±0.6	4.6±0.6	3.4±0.0	56.2±1.8
Overall mean	8	160.9±32.2	-24.0±1.1	8.3±0.5	3.8±0.2	3.4±0.0	54.8±1.7

Note: TPOS, trophic position; LGTH, length; SHTC, Swan Hills Treatment Centre.

Appendix A7. Summary data for MeHg concentration, stable isotope ratios, trophic position, and body length: *Physa* spp.

Lake	N	$\delta^{13}C$ (‰)	$\delta^{15}N$ (‰)	LGTH (mm)
Burned lakes				
Mons	6	-23.4±1.1	2.7±0.2	10.7±0.1
Twin	7	-27.4±0.2	2.1±0.1	12.1±0.1
Shelly	7	-24.7±0.8	1.9±0.5	12.7±0.0
Michel	7	-27.2±0.3	2.2±0.3	13.9±0.0
Swartz	7	-29.5±0.4	3.0±0.2	10.7±0.0
Mean (burned)	5	-26.4±1.1	2.4±0.2	12.0±0.6
Reference lakes				
Rat	7	-23.3±1.0	1.9±0.3	13.0±0.0
Marigold	5	-20.5±0.2	2.7±0.2	
Pleasant	7	-16.3±0.9	1.0±0.3	13.0±0.1
Goodwin	7	-28.1±0.6	2.8±0.1	13.9±0.1
Delorme	7	-22.6±0.3	3.6±0.2	13.7±0.1
Mean (reference)	5	-22.1±1.9	2.4±0.4	13.4±0.2
SWT lakes				
Edith	4	-23.9±0.1	3.1±0.1	13.1±0.0
Christina	7	-26.4±0.4	4.3±0.2	12.6±0.0
SWT mean	2	-25.1±1.2	3.7±0.6	12.8±0.2
Overall mean	12	-24.4±1.1	2.6±0.2	12.7±0.3

Appendix B:

Pearson correlation coefficients for MeHg concentrations and environmental variables

Table B-1. Pearson correlation coefficients for MeHg concentrations and environmental variables

Environmental variables	MeHg (ng·g dry wt $^{-1}$)											
	Trichoptera $n = 10$		*G. lacustris* $n = 7$		*C. shurtleffi* $n = 8$		*Aeshna* spp. $n = 9$		*Notonecta* spp. $n = 12$		*C. inconstans* $n = 8$	
	r	*P*	*r*	*P*	*r*	*P*	*r*	*P*	*r*	*P*	*r*	*P*
LA (ha)	0.06	0.87	-0.37	0.42	0.06	0.88	0.24	0.53	-0.04	0.89	-0.01	0.98
Elevation (m)	0.43	0.21	0.66	0.11	0.49	0.22	0.48	0.18	0.23	0.47	0.55	0.16
CA/LV (m^{-1})	-0.20	0.78	0.13	0.78	-0.44	0.27	-0.54	0.12	-0.24	0.46	-0.49	0.22
%CON	0.73	0.02	0.86	0.01	0.66	0.08	0.60	0.09	-0.16	0.62	0.68	0.06
%WTLD	-0.15	0.68	0.22	0.64	0.46	0.26	0.87	0.002	-0.09	0.79	0.67	0.07
Conductivity ($\mu S \cdot cm^{-1}$)	-0.60	0.07	-0.48	0.27	-0.67	0.07	-0.48	0.20	-0.24	0.46	-0.62	0.10
pH	-0.42	0.22	-0.48	0.28	-0.90	0.002	-0.69	0.04	-0.44	0.16	-0.86	0.007
TP ($\mu g \cdot L^{-1}$)	-0.48	0.16	-0.57	0.18	-0.80	0.02	-0.84	0.004	-0.38	0.22	-0.78	0.02
TDP ($\mu g \cdot L^{-1}$)	-0.16	0.66	-0.18	0.70	-0.45	0.26	-0.51	0.16	-0.65	0.02	-0.47	0.24
Colour (mg Pt·L^{-1})	0.11	0.75	0.06	0.90	-0.08	0.85	0.03	0.95	-0.26	0.42	-0.007	0.99
DOC ($mg \cdot L^{-1}$)	-0.40	0.25	-0.45	0.31	-0.57	0.14	-0.38	0.31	-0.58	0.05	-0.46	0.25
Chl *a* ($\mu g \cdot L^{-1}$)	-0.51	0.14	-0.67	0.10	-0.63	0.09	-0.73	0.03	-0.19	0.55	-0.68	0.06
ALK ($mg \cdot L^{-1}$)	-0.52	0.12	-0.43	0.33	-0.76	0.03	-0.62	0.07	-0.27	0.39	-0.74	0.03
DIN ($\mu g \cdot L^{-1}$)	-0.22	0.54	-0.15	0.75	-0.12	0.77	0.41	0.28	-0.007	0.98	0.08	0.84
Ca^{2+} ($mg \cdot L^{-1}$)	-0.45	0.19	-0.22	0.63	-0.75	0.03	-0.53	0.14	-0.25	0.42	-0.67	0.07
K^{+} ($mg \cdot L^{-1}$)	-0.53	0.12	-0.90	0.001	-0.40	0.33	-0.55	0,12	-0.004	0.99	-0.56	0.14
SO_4^{-2} ($\mu g \cdot L^{-1}$)	0.23	0.52	0.46	0.3	-0.45	0.3	-0.45	0.26	-0.21	0.50	-0.33	0.41

Note: LA, lake area; CA/LV, catchment area/lake volume;%CON, conifer cover; %WTLD, wetland cover; TP, total phosphorus; TDP, total dissolved phosphorus; DOC, dissolved organic carbon; Chl *a*, chlorophyll *a*; ALK, alkalinity; DIN, dissolved inorganic nitrogen.

Appendix C:

Linear regressions of $\log_{10}$MeHg vs. $\delta^{15}N$ in aquatic macroinvertebrates

Table C-1. Linear regressions of log_{10}MeHg vs. $\delta^{15}N$ in aquatic macroinvertebrates

Lake	Regression equation	r^2	P
Mons	$Log_{10}MeHg = 1.02\pm0.07 + 0.21\pm0.02*\delta^{15}N$	0.78	<<0.01
Twin[a]	$Log_{10}MeHg = 0.95\pm0.12 + 0.32\pm0.04*\delta^{15}N$	0.67	<<0.01
Shelly	$Log_{10}MeHg = 1.34\pm0.03 + 0.21\pm0.01*\delta^{15}N$	0.86	<<0.01
Michel[a]	$Log_{10}MeHg = 1.16\pm0.12 + 0.23\pm0.04*\delta^{15}N$	0.64	<<0.01
Swartz	$Log_{10}MeHg = 1.01\pm0.53 + 0.20\pm0.13*\delta^{15}N$	0.11	0.13
Rat	$Log_{10}MeHg = 1.19\pm0.04 + 0.24\pm0.02*\delta^{15}N$	0.85	<<0.01
Marigold[a]	$Log_{10}MeHg = 0.56\pm0.15 + 0.37\pm0.04*\delta^{15}N$	0.78	<<0.01
Pleasant	$Log_{10}MeHg = 1.31\pm0.04 + 0.17\pm0.02*\delta^{15}N$	0.76	<<0.01
Goodwin	$Log_{10}MeHg = 1.18\pm0.04 + 0.20\pm0.02*\delta^{15}N$	0.80	<<0.01
Delormie	$Log_{10}MeHg = 1.06\pm0.06 + 0.13\pm0.02*\delta^{15}N$	0.60	<<0.01
Edith[b]	$Log_{10}MeHg = 1.64\pm0.44 + 0.12\pm0.10*\delta^{15}N$	0.07	0.25
Chrystina[a]	$Log_{10}MeHg = 3.05\pm0.14 - 0.17\pm0.03*\delta^{15}N$	0.72	<<0.01

Note: [a]Trichoptera, Odonates, and *Notonecta* spp. only; [b]Odonates and *Notonecta* spp. only.